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Impacts on Quality of Inland Wetlands of the United States: A Survey of Indicators, Techniques, and Applications of Community-Level Biomonitoring Data

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IMPACTS ON QUALITY OF INLAND WETLANDS OF THE UNITED STATES:

A SURVEY OF INDICATORS, TECHNIQUES, AND APPLICATIONS OF COMMUNITY-LEVEL BIOMONITORING DATA

by:

Paul R. Adamus
NSI Technology Services Corporation
US EPA Environmental Research Laboratory
200 SW 35th St.
Corvallis, OR 97333

and

Karla Brandt
Center for Wetlands
University of Florida
Gainesville, FL 32611

Eric M. Preston, Project Officer
USEPA Environmental Research Laboratory
200 SW 35th St.
Corvallis, Oregon 97333

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This report focuses on inland wetlands of the conterminous United States. Except for those bordering the Great Lakes, these are not subject to significant tidal fluctuations. They are generally fresh water wetlands, except for saline wetlands in some mid-continent and western regions. Other tidal, tundra, and tropical wetlands were not included because their consideration would have involved a greatly expanded scope of work. Protocols for biological sampling of tidal wetlands have been presented by Simenstad et al. and others. For purposes of this report, "wetlands" are considered to be vegetated areas transitional between uplands and open water.

A principal goal of this report is to encourage each state to track their progress in protecting wetland ecological condition. It emphasizes the biological functions of wetlands - habitat for fish, wildlife, and related organisms and the processes that support biological functions.

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SUMMARY

This report describes what is known about ecological community response to anthropogenic stressors in inland wetlands. Because wetlands are shallow, located in a topographically low position in the landscape, and have low hydraulic exchange rates, they are particularly sensitive to accumulation of pollutants and changes in water tables. Despite this situation, and the fact that unimpacted wetlands support exceptional biological production, government biomonitoring programs to date have focused mainly on rivers and lakes to the exclusion of wetlands. Monitoring of wetlands has focused mainly on extent of the resource, rather than changes in wetland quality (e.g., ecological structure and function, condition).

Based on a synopsis of the literature, the report describes the potential effects upon wetland community structure of the following stressors: eutrophication, organic loading, contaminant toxicity, acidification, salinization, sedimentation, turbidity/shade, vegetation removal, thermal alteration, dehydration, inundation, and fragmentation of habitat. The incidence and geographic extent of these stressors in wetlands is currently unknown. Information is provided concerning the effect of each stressor on potential indicators of wetland condition--wetland microbes, algae, vascular plants, invertebrates, fish, amphibians, reptiles, birds, mammals, and selected biological processes in wetlands.

The report describes options for using potential indicators to (a) develop and incorporate biocriteria for the protection of sustainable ecological conditions, and (b) help identify and prioritize degraded wetlands that may be candidates for restoration. Because of the lack of appropriate comparative studies of wetlands, the report does not provide biocriteria for wetlands, evaluate or prioritize potential indicators of wetland condition, nor endorse specific techniques for wetland biomonitoring and data analysis. Its intended use is mainly as a technical source document for future design, testing, and reporting of indicators.

The focus is primarily on community-level (as opposed to individual-organism) responses to the stressors. Techniques for sampling each of the taxonomic groups in wetlands are described generally. To the extent allowed by published data, the range of density, richness, and diversity within some taxonomic groups is reported, and most-sensitive species are noted. To facilitate regionalization of future efforts and to further cooperation among researchers and use/analysis of extant data, the locations of a large portion of published wetland community studies are depicted on state maps, referenced to state bibliographies. Important elements in future use and regionalization of this report's information should be continued reviews by other scientists of literature published after 1989, and expanded compilations of existing data on responses of individual species.

Copies of this report are available from:

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1.0 INTRODUCTION

The widespread physical loss of North American wetlands has been generally documented (e.g., Tiner 1984). However, uncertainty exists regarding the ecological condition of the wetlands that remain. Although wetlands passively provide for many public uses--e.g., water purification, flood control, aquatic life and wildlife support--the extent to which these functions are being impaired in the remaining wetland resources is unclear. The Environmental Protection Agency (EPA) has the responsibility under several legal mandates (Table 1) for determining this.

Wetland ecological quality assumes special significance because of current State and Federal interest in adopting a policy of "no net loss" of the nation's wetlands. As expressed by EPA's Wetlands Action Plan, this implies no net loss of either acreage or **function**. To determine whether particular functions or uses, such as support of aquatic life, are being impaired in wetlands, "indicators" of these functions must be identified and protocols articulated for their measurement and interpretation.

1.1 SCOPE OF COVERAGE and ORGANIZATION

This report focuses on **inland wetlands of the conterminous United States.** Except for those bordering the Great Lakes, these are not subject to significant tidal fluctuations. They are generally fresh water wetlands, except for saline wetlands in some mid-continent and western regions. **Other tidal, tundra, and tropical wetlands were not included** because their consideration would have involved a greatly expanded scope of work. Protocols for biological sampling of tidal wetlands have been presented by Simenstad et al. (1989) and others. For purposes of this report, "wetlands" are considered to be vegetated areas transitional between uplands and open water.

A principal goal of this report is to encourage each state to track their progress in protecting wetland ecological condition. As one of many components needed to achieve this, this report identifies data gaps and provides guidance that describes (a) how existing resource data might be applied in the designation of "uses" for wetlands, (b) ambient **biological** criteria for wetlands might be developed or modified, and (c) how wetlands might be periodically sampled (and data interpreted) to estimate their relative ecological condition, compliance with biological criteria, or need for restoration. Publication of this report is not intended to imply that sufficient knowledge exists to develop community-based biocriteria for all wetlands at the present time.

This report emphasizes the biological functions of wetlands--habitat for fish, wildlife, and related organisms and the processes that support biological functions. Its purpose is to provide State and Federal water quality and wetland managers with a synopsis of selected literature describing the community-level response of wetlands and similar aquatic systems to particular stressors. In most cases, **this document does not synthesize the literature into statements applicable to all wetlands, or to all wetlands, taxa, or stressors of a certain type.** Such a synthesis was generally avoided because the technical literature lacks a sufficient number of studies that demonstrate causal relationships (as opposed to correlation) or that allow statistical extrapolation (i.e., synthesis) to entire taxa, stressor types, or wetland types, regions, or states.

Biological sampling can be carried out at several ecological levels--the organism, the population, the community, or the ecosystem (Table 2). This report focuses on measurements of **biological communities**, that is, associations of interacting populations, usually delimited by their interactions or by spatial occurrence. Tables 3 and 4 show specific metrics (that is, characteristics or indices) used to describe the communities. This report also discusses, to a more limited extent, the measurement and use of biological processes as indicators of anthropogenic stress.

Table 1. Examples of Major Federal Laws, Directives, and Regulations for the Management and Protection of Wetlands.

Directive	Date	Responsible Agency
Executive Order 11990 Protection of Wetlands	May 1977	All agencies
Executive Order 11988 Floodplain Management	May 1977	All agencies
Federal Water Pollution Control Act (PL 92-500) as Amended	1972, 1977	
Section 401- Water Quality Certification		EPA, States
Section 404- Dredge and Fill Permit Program		EPA, Corps of Engineers
reporting requirements for Section 305(b)		States
National Environmental Policy Act	1975	All agencies
Coastal Zone Management Act	1972	Office of Coastal Zone Management

This report's focus on biological communities does not mean other measurements are less important or useful. Indeed, there are numerous situations where alternative indicators--in particular, wetland flooding regime, bioaccumulation of contaminants, sedimentation rate, population demographics, and habitat structure --can more cost-effectively reflect the ecological condition, impact causes, and sustainability of a wetland than can community-level biological methods. Quantitative literature on the community ecology of wetlands has been singled out for focus, largely because of current EPA interest in applying this approach when assisting States with the development of community-based biocriteria for surface waters (Plafkin et al. 1989, USEPA 1987, 1990).

This focus on community-level measurements coincides with a growing body of literature which suggests that, at least for many applications in flowing waters, monitoring of biological community structure provides cost-effective information about ecological condition or as some have termed it, "health" (Krueger et al. 1988). Biological monitoring directly addresses the result of pollution, not its possible cause. Measurements of community structure can integrate intermittent stressor conditions. They can also detect impacts from many sources for which chemical criteria are poorly suited to detect (e.g., alteration of hydrologic regimes, synergistic pollutant effects, nonpoint runoff). If community-level measurements suggest that a stress is occurring, traditional methods (e.g., direct hydrologic monitoring, tissue analysis, chemical sampling) can be

Table 2. Potential Metrics for Wetland Biomonitoring.

Organismal Level

Altered Behavioral Responses
- o foraging/feeding effectiveness
- o response to odors, pheromones, temperature, chemicals
- o reproductive behavior (courtship, mating, maternal/paternal)
- o predator avoidance (reaction time, evasiveness)
- o migratory/dispersal behavior
- o social interactions/territoriality

Altered Metabolism/Homeostasis
- o thermo/osmo/hydro regulation
- o oxygen consumption, photosynthesis
- o nutrient uptake and translocation, food conversion efficiency
- o enzyme/protein activation/inhibition (e.g., cholinesterase)
- o hormone balances

Altered Reproductive Success
- o seed set, tillering, flowering, vegetative (clonal) growth
- o sexual maturity, conception/implantation, parturition

Altered Growth and Development
- o growth rate (e.g., tree ring analysis)
- o size at age, morphological abnormalities

Decreased Disease Resistance
Direct Tissue/Organ Damage (e.g., lesions, tumors)
Changes in Stamina (e.g., plant vigor)
Bioaccumulation

Population Level
- o survival/mortality
- o sex ratio, fecundity
- o population abundance, biomass, density
- o age structure and recruitment
- o gene pool
- o intraspecific competition
- o population behavior, migration, dispersal
- o susceptibility to predation
- o population rate of decline or increase

Community Level (see Table 3 for details)
Structure (taxonomic and functional)
Function (process)

Ecosystem Level
Mass Balance of Nutrients

Table 3. Examples of Biological Metrics Describing Wetland Community Structure and Function.

Community Structure

Abundance. The number of individuals of an organism or organisms. As an analytic metric, tends to exaggerate the importance of small, abundant species.

Biomass. The weight of living material in all or part of a community. For this report, it includes measurements of chlorophyll or caloric content as well. As an analytic metric, tends to exaggerate the importance of large, uncommon species.

Density. The number of individuals of an organism or organisms, per unit area or per unit volume.

Richness. The number of species, size classes, or other functional groups, per unit area or volume, or per number of individuals.

Diversity. The variety (richness) of species, life forms (**physiognomy**), genetic material, or functional groups, taking into account the relative abundance (**evenness and dominance**) of each species or group.

Community Composition. Qualitative descriptions of the members of a community (e.g., species lists), perhaps describing as well their relative abundance and grouped by their attributes (e.g., exotic vs. native, migrant vs. resident, response guild).

Community Attributes

Colonization rates

Stability
- o resistance, assimilation capacity
- o resilience, recovery rate

Successional relationships
Food web structure, trophic interactions
Competition among species
Predator/prey relationships
Grazing/herbivory relationships
Parasite/host relationships, symbiosis

Community Function (Process)

Decomposition/leaching
Productivity, Photosynthesis, Respiration
Denitrification, Nitrogen Fixation
Other Biogeochemical Functions (e.g., methanogenesis)

Table 4. Examples of Analytical Metrics, Indices, and Procedures Used for Wetland Community Studies.

Similarity (Comparative) Indices. Metrics that reflect the number of species or functional groups in common between multiple wetlands or time periods. May be weighted by relative abundance, biomass, taxonomic dissimilarity, or caloric content of the component species. Includes Jaccard coefficient, Bray-Curtis coefficient, rank coefficients, overlap indices, the "community degradation index" (Ramm 1988), and others.

Cluster Analysis and Ordination. Procedures that detect statistical patterns and associations in community data. Can be used to hypothesize relationships to a stressor. Includes principal components analysis, reciprocal averaging, detrended correspondence analysis, TWINSPAN, canonical correlation, and others. Can be used to identify guilds (see below). A useful reference is Pielou (1984), and a cautionary note is expressed by Beals (1973).

Food Web Analysis. Procedures that measure length of food chains, number of trophic levels, ratio of number of trophic species to trophic links, and similar measures (e.g., Patten et al. 1989, Turner 1988). As yet, they have seldom been tested in stressed wetlands.

Tolerance Indices. Metrics that reflect proportionate composition of tolerant vs. intolerant taxa. Includes saprobic indices, macroinvertebrate EPT index, Hilsenhoff index, and others detailed and compared in Hellawell (1984) and Washington (1984). "Tolerance" usually means tolerance to organic pollution; tolerance to many toxicants and physical habitat alterations may not be well-reflected by available indices.

Guild Analysis. Procedures in which individual species are assigned to functional groups (species assemblages) based on similar facets of their:

- life history
- habitat preference
- trophic level, assumed niche breadth
- size, biomass, caloric content
- toxicological sensitivity
- behavioral characteristics
- phenological characteristics
- sensitivity to human presence
- status as an exotic or indigenous species
- resident vs. migrant status
- harvested vs. protected status
- other factors

Indices of Biotic Integrity. Indices that are a composite of weighted metrics describing richness, pollution-tolerance, trophic levels, abundance, hybridization, and deformities. Widely used in stream fish studies (see Karr 1981).

used to help determine cause. Moreover, ambient biological criteria can directly provide realistic evaluations of whether specific areas designated for protection of aquatic life are meeting this objective, or require restoration.

In most cases, if biological community monitoring data are to be correctly interpreted, they should be collected over time periods spanning several years, and should be accompanied with hydrologic and water quality measurements. Hydrologic measurements typically describe the variability, temporal pattern, extent, frequency, depth, and duration of surface waters and/or saturated condition (e.g., Gunderson 1989, Poff and Ward 1989). They may be expressed, for example, as water residence time distribution, water yield (net water balance), and stage or flow exceedence curves (i.e., percentage of time a particular water level or discharge is exceeded). Typical equipment for measuring these includes precipitation gauges, flourescent dyes, stage-discharge recorders, piezometers, redox probes, and sediment traps. For further information on the use of hydrologic and sediment measurements in wetland monitoring, readers may find the following references particularly useful:

> Faulkner et al. 1989, Gunderson 1989, Heliotis and DeWitt 1987, Kadlec 1984, Kadlec 1988, LaBaugh 1986, Rosenberry 1990, USEPA 1985, Van Haveren 1986, Welcomme 1979, and Zimmerman 1988.

Monitoring protocols for estimating bioaccumulation in wetlands will be published in a manual by the U.S. Fish and Wildlife Service in 1990. General summaries of aquatic bioaccumulation processes and effects are contained in Biddinger and Gloss 1984, Fagerstrom 1979, Phillips 1980, Robinson-Wilson 1981, and Sonstegard 1977. Examples of bioaccumulation studies in wetlands include:

> Anthony and Kozlowski 1982, Aulio 1980, Behan et al. 1979, Lambing et al. 1988, Larsen and Schierup 1981, McIntosh et al. 1978, Metcalf et al. 1984, Mouvet 1985, Niethammer et al. 1985, Schierup and Larsen 1981, Stephenson and Mackie 1988, Taylor and Crowder 1983, and others.

It is assumed that each state will determine how best to sample wetlands, incorporate wetland biological criteria into its water quality management programs, and establish restoration priorities. For this reason, much of the information contained in this report is presented as "could's" or "might's," and details regarding "how" the many technical statements should be interpreted and implemented are left to other agencies and institutions which have diverse goals and which encounter a wide variety of political and environmental conditions. To date, only a single state (Florida) has drafted survey-based biological criteria for some of its wetland resource (described by Schwarz 1987).

This report is also intended to serve as once source of technical support for the EPA's National Guidance on Water Quality Standards for Wetlands, prepared jointly by the Office of Wetlands Protection and the Office of Water Regulations and Standards. This report pursues this goal partly by providing just one input--a literature review--for identifying and interpreting biological indicators of wetland ecological condition.

Many factors other than technical data must be considered in developing biological criteria and setting restoration priorities. Decisions concerning selection of which resources, uses, or functions to protect or enhance are inevitably complex, since the criteria for protecting one resource or use may be counter to protecting another (Duinker and Beanlands 1986, Graul and Miller 1984, Smith and Theberge 1987). A generalized list of wetland functions or uses that might be the focus of protection or restoration is contained in Section 404 of the Clean Water Act. These are as follows (from 33 CFR 320 (b)(2)):

a. Food Chain Production (i)
b. General Habitat (i)
c. Research, Education, and Refuges (ii)
d. Hydrologic Modification (iii)
e. Sediment Modification (iii)

f. Wave Buffering and Erosion Control (iv)
g. Flood Storage (v)
h. Ground Water Recharge or Discharge (vi)
i. Water Purification (vii)
j. Uniqueness/Scarcity (viii)

Any such list could include many additional or more specific values of wetlands, e.g., maintenance of biodiversity, landscape value as corridors or habitat islands, role in global climate warming, timber harvest.

This report begins, in Section 2, with a description of possible technical approaches that state and local agencies might use in designating "uses" for wetlands and eventually, perhaps, developing community-level biological criteria. Section 3 then describes general considerations in the design of wetland biomonitoring studies. Remaining sections of the report are delimited by major taxonomic groups (e.g., birds, fish). Each of these taxonomic sections is divided according to the following themes:

Use as Indicators
Sampling Protocols and Equipment
Spatial and Temporal Variability, Data Gaps

Originally, our intent was to organize the discussions by **wetland type**. This is because wetland types are generally believed to differ in their community-level responses to particular stressors. Thus, wetland "type" may be an important qualifier of any biocriteria that might be developed in the future. However, studies of specific anthropogenic stressors within individual types of wetlands were often so few that attempts to organize sections by wetland type proved futile. Nonetheless, within the discussions of particular taxa and stressors, statements about indicator metrics and taxa have been couched whenever possible in terms descriptive of wetland type/region. Also, attempts were made to organize the descriptions of sampling techniques according to wetland type. Although sampling protocols and appropriate equipment differ between flowing-water wetlands, wetlands with standing surface water, and wetlands without surface water, a finer classification of types is difficult to specify without knowledge of study objectives. Usually, having a clear definition of the objectives of a particular study is more important to study design than is knowledge the particular types of wetlands that happen to be included in a study.

The subsection discussions of **Use as Indicators** attempt to document community-level shifts that occur as a result of particular anthropogenic stressors. Stressors considered in this report are listed and defined in Table 5. Their effects on biota are often cumulative and interactive, thus complicating the use of biota as indicators of any individual stressor. Although several previous documents have summarized impacts to wetland biota (e.g., Brennen 1985, Brown et al. 1989, Darnell et al. 1976, Davis and Brinson 1980, USEPA 1983, USEPA and USFWS 1984), not all taxa, wetland, and stressor types have been covered and inferences have commonly been drawn from non-wetland aquatic environments.

It is important to understand that statements made in this report reflect strongly the particular wetland locations and types that were studied, and considerable uncertainty exists regarding whether such conclusions (e.g., about the value of specific taxa as indicators) can be transferred to other wetland types and regions. Many cited studies reflect one-visit or one-season data collections from a single wetland type, rather than recurrent monitoring. There are very few statistically-valid studies that adequately quantify the exposures of wetland organisms to stressors using factorial designs, e.g., studying areas both with/without treatment and staggered before/after measurements (Stewart-Oaten et al. 1986, Walters et al. 1988). Although it may appear, from the quantity of studies contained in the maps, in their bibliographies, and in the extensive literature citations in the text that inland wetlands in some regions have been extensively monitored, in truth relatively little is known about wetland biological response to anthropogenic stressors. Compared to monitoring of streams and lakes, sampling of wetlands on a recurrent or comparative regional basis has been almost non-existent, partly due to lack of government sponsorship of wetland biomonitoring programs.

Also, the response of a wetland community to anthropogenic stress depends not only on the taxa present and the severity of the stressor, but also on the geomorphic, physical, and chemical environment of the wetlands (Adamus et al. 1987, Adamus and Stockwell 1983). For example:

- Wetland biological communities most vulnerable to sedimentation effects might be those located in shallow basins without outlets, so sediment quickly accumulates;

- Wetland biological communities most vulnerable to eutrophication and contaminant effects might be those in wetlands that get most of their water directly from precipitation (e.g., ombrotrophic bogs), which have low alkalinity, and/or which have types of sediments that adsorb (but do not render biologically unavailable or harmless) the nutrients and contaminants during the short time that runoff passes through the wetland, e.g. Goldsborough and Beck (1989).

- Wetland biological communities most vulnerable to effects of many anthropogenic changes might be those that:

    (a) have no prior exposure to similar levels or types of stress; and/or
    (b) exist in wetland types or regions that are characteristically stable (relatively speaking) over time; and/or
    (c) are physically isolated from sources of colonizers, so that recovery occurs slowly; and/or
    (d) are located in regions that have experienced especially rapid losses of wetlands of a similar type.

Considerably more investigation may be required before candidate indicators of wetland ecological condition can be fairly rated relative to one another, and exact numerical criteria specified. Thus, **users of the report are urged to obtain, whenever possible, assistance from local wetland scientists when attempting to apply the information reported herein.**

In this report, the representativeness, replication, and field and data analysis techniques used by cited studies were not evaluated; the overwhelming majority of citations are peer-reviewed papers from professional journals. **Also, no attempt is made to give equal coverage to all topics within the general theme, because availability of data varies greatly among topics.**

In the "Use as Indicators" subsections, discussions focus on the community metrics that are defined in Table 4. An important metric that is frequently discussed is "richness." References in this report to the response of richness to stressors should be assumed, unless otherwise noted, to refer to changes in taxonomic richness within a wetland. However, readers should be aware that some stressors may increase richness of a major taxonomic group within a wetland (alpha diversity) while decreasing richness on a regional level (beta and gamma diversity). This may occur as the result of a net increase in species within the wetland, but an increase in which regionally rare species originally inhabiting the wetland are replaced by a larger number of regionally common and widespread species. Thus, no value judgement should necessarily be attached to statements that richness increases in response to a stressor. Moreover, design of future studies evaluating changes in community richness should in many cases include information on the regional rarity of species that may be displaced.

The subsection discussions of **Sampling Protocols and Equipment** focus on techniques for sampling each taxonomic group, e.g., how, where, when, and how often sampling has been done. However, **this report is not intended to be a prescriptive manual.** Rather, the intent is to present the user with choices. Choices are provided by summarizing the types of equipment, protocols, and community metrics that have been used previously to monitor wetland communities. Choices, rather than prescriptions, are given because rigid

Table 5. Stressors Addressed in This Report.

Enrichment/Eutrophication. Increases in concentration or availability of nitrogen and phosphorus. Typically associated with fertilizer application, cattle, ineffective wastewater treatment systems, fossil fuel combustion, urban runoff, and other sources.

Organic Loading and Reduced DO. Increases in carbon, to the point where an increased biological oxygen demand reduces dissolve oxygen in sediments and the water column and increases toxic gases (e.g., hydrogen sulfide, ammonia). Typically associated with ineffective wastewater treatment systems.

Contaminant Toxicity. Increases in concentration, availability, and/or toxicity of metals and synthetic organic substances. Typically associated with agriculture (pesticide applications), aquatic weed control, mining, urban runoff, landfills, hazardous waste sites, fossil fuel combustion, wastewater treatment systems, and other sources.

Acidification. Increases in acidity (decreases in pH). Typically associated with mining and fossil fuel combustion.

Salinization. Increases in dissolved salts, particularly chloride, and related parameters such as conductivity and alkalinity. Typically associated with road salt used for winter ice control, irrigation return waters, seawater intrusion (e.g., due to land loss or aquifer exploitation), and domestic/industrial wastes.

Sedimentation/Burial. Increases in deposited sediments, resulting in partial or complete burial of organisms and alteration of substrate. Typically associated with agriculture, disturbance of stream flow regimes, urban runoff, ineffective wastewater treatment plants, deposition of dredged or other fill material, and erosion from mining and construction sites.

Turbidity/Shade. Reductions in solar penetration of waters as a result of blockage by suspended sediments and/or overstory vegetation or other physical obstructions. Typically associated with agriculture, disturbance of stream flow regimes, urban runoff, ineffective wastewater treatment plants, and erosion from mining and construction sites, as well as from natural succession, placement of bridges and other structures, and resuspension by fish (e.g., common carp) and wind.

Vegetation Removal. Defoliation and possibly reduction of vegetation through physical removal, with concomitant increases in solar radiation. Typically associated with aquatic weed control, agricultural and silvicultural activities, channelization, bank stabilization, urban development, defoliation from airborne contaminants and other stressors included in this report, grazing/herbivory (e.g., from muskrat, grass carp, geese, crayfish, insects), disease, and fire.

Thermal Alteration. Long-term changes (especially increases) in temperature of water or sediment. Typically associated with power plants, other industrial facilities, and global climate warming.

Dehydration. Reductions in wetland water levels and/or increased frequency, duration, or extent of desiccation of wetland sediments. Typically associated with ditching, channelization of nearby streams, invasion of wetlands by highly transpirative plant species, outlet widening, subsurface drainage, global climate change, and ground or surface water withdrawals for agricultural, industrial, or residential use.

Inundation. Increases in wetland water levels and/or increase in the frequency, duration, or extent of saturation of wetland sediments. Typically associated with impoundment (e.g., for cranberry or rice cultivation, flood control, water supply, waterfowl management) or changes in watershed land use that result in more runoff being provided to wetlands.

Fragmentation of Habitat. Increases in the distance between, and reduction in sizes of, patches of suitable habitat.

Other Human Presence. Increases in noise, predation from pets, disturbance from visitation, invasion by aggressive species capable of outcompeting species that normally characterize intact communities; electromagnetic, ultraviolet (UV-B), and other radiation, and other factors not addressed above.

standardization of wetland monitoring techniques may not be desirable or feasible given the current lack of comparative studies. Exceptions may include situations where litigation is probable or efficacy of a regulatory program must be determined. Also, the need for diverse and adaptive sampling stategies is suggested by the extreme temporal and spatial variability within and among wetlands, and the variety of purposes for which wetlands are monitored.

The Sampling Protocols subsection also notes where data are indicate that one protocol, type of sampler, or metric is better than another. However, we have not evaluated these ourselves, except to note situations where the use of a particular sampler, protocol, or community metric seems clearly inappropriate.

Finally, the subsection discussions of **Spatial and Temporal Variability--Data Gaps** summarize numerical data, both temporal and spatial, on wetland community ecology. The range in values of, say, macroinvertebrate density, is noted for wetland types for which such data are available.

Data on variability is potentially useful for helping develop wetland biocriteria. For example, taxa whose community structure naturally varies the least with time and space tend to be most practical for use as indicators of anthropogenic influences. Also, the spatial variability in community composition among wetlands may be less in disturbed landscapes than in natural landscapes, if inferences from other ecosystem types are applicable (Sheehan 1984). Such information is useful in design of regional monitoring programs.

If data that describe variability were drawn from a sufficient number of wetlands to represent the wetland resource of a region, and with a sufficient frequency to capture the range of changing conditions, then such data might be used as one basis for establishing numeric criteria for protection of wetland aquatic life. They might also be used to target gaps and reduce costs in the statistical design of more rigorous biomonitoring efforts. Such an approach has been proposed for use in EPA's new Environmental Monitoring and Assessment Program (EMAP), and has been applied successfully to stream ecosystems in Ohio and Arkansas (e.g., Giese et al. 1987).

However, existing data, such as those presented, are of uncertain statistical representativeness. They were compiled from all relevant, published studies. As such, they may represent only a first-guess or "default" estimate of expected or baseline levels of community-level metrics, relevant only when local data are lacking. As noted earlier, conclusions drawn from these data cannot be extrapolated to other wetlands with known certainty.

Areas of missing biological information ("data gaps") are also noted. As appropriate, gaps are identified by geographic region, by wetland type, and by type of stressor. Emphasis is on geographic gaps, rather than on thematic gaps (thematic gaps have been identified in Adamus 1989, USEPA 1988, and in many other documents). Information on gaps was gained partly by plotting all relevant studies on state maps (Appendix B).

1.2 HOW THE REPORT WAS PREPARED

In August 1988, EPA's Wetlands Research Program sponsored a workshop in Easton, Maryland, a part of which focused on identifying organisms and metrics that might be useful for indicating wetland ecological condition. Findings were summarized in an EPA report , "Wetlands and Water Quality: EPA's Research and Monitoring Implementation Plan for the Years 1989 - 1994" (Adamus 1989). That report noted a need for synthesizing existing regional literature in ways that would allow candidate bioindicators to be identified and available data to be numerically compiled. Potential categories of indicators applicable to surface waters (in general) were targeted in EPA contracted reports (Mittleman et al. 1987) and by another EPA workshop held in early 1989 (Temple, Barker, & Sloane 1989).

A preliminary synthesis of wetland indicator literature was completed in August 1989 (Brown et al., unpublished) as part of EPA's planning efforts for the new Environmental Monitoring and Assessment Program (EMAP). That effort included a review of abiotic as well as biotic indicators, but did not attempt a comprehensive review of technical literature. At the same time, EPA's Office of Policy, Planning, and Evaluation (OPPE), acting on a request from EPA's Office of Wetlands Protection, asked EPA's Corvallis Environmental Research Laboratory to modify and expand the scope of the similar, unpublished EMAP report. Representatives of EPA's Office of Water Regulations and Standards (OWRS) were also involved in early discussions of the scope of the effort. It was agreed that the modified report would focus more strongly than did the EMAP effort on compiling quantitative measurements of wetland ecological communities. In particular, it would attempt to describe the variability in community responses by region, stressor type, and taxon. Additional support from EMAP would complement OPPE's support. This report represents that effort.

Literature review began with an automated bibliographic search of the Wetland Values Database of the U.S. Fish and Wildlife (Ruta Stuber 1986). Other bibliographic databases were also searched using terms wholly or partly synonymous with wetlands, e.g.:

> alluvial; aquatic moss; aquic; aquod; backwater; bayou; benthic/aquatic/submersed/submerged plant/vegetation; black(-)water; bog; bosque; brown(-)water; depression; ditch; dystroph-; fen; floodplain; fluventic; fluvisol; histic; histosol; hydrophy-; intermit- stream; inundated soil; lagoon; lentic; littoral; lowland; macrophy-; marsh; mire; muck; muskeg; oxbow; playa; pluvial; pocosin; pond; poorly drained; pothole; riparian; saprophilic; seep; shallow lake; shoal; sphag-; stockpond; stream corridor; swamp; vernal pool; wash; water log-; wet land; wet meadow; wet prairie

Literature was included if it met the following criteria:

o quantitative biological measurements were described (i.e., not just species lists or faunistic surveys);

o inland nontidal wet areas were covered;

o oriented towards the community level of ecological structure (i.e., transects or point data in which a full range of vascular plant, fish, bird, amphibian, or mammal species was measured, not just single species);

o if not community-oriented, then focused on the sensitivity of ecosystem process (e.g., productivity, decomposition) to environmental stressors, or on the relative usefulness of particular species as "bioindicators."

References resulting from the preliminary literature review were compiled by state and circulated, with the criteria, for comment to persons from the following groups:

o wetland coordinators from the EPA Regions and wetland biologists from other EPA Labs
o selected offices of the Corps of Engineers in each region
o a majority of members of the Society of Wetland Scientists
o wetland coordinators for all state highway departments
o state biologists of the Soil Conservation Service
o refuge managers of the National Wildlife Refuges
o attendees from the Easton workshop
o other persons selected from Wetland Research Program mailing lists

In addition to soliciting comments on published literature, we asked these persons to suggest data meeting our criteria that could be found in the following types of less-available literature:

- student theses
- biomonitored mitigation sites
- impact statements or permit applications
- Superfund site assessments
- water quality bioassessment reports
- utility siting plans
- fish/wildlife agency studies
- forest management monitoring plans
- grazing management monitoring plans
- aquatic weed control impact studies

A large number of responses were received, and along with secondary citations discovered in literature we collected, resulted in significant expansion of our bibliography. Some unpublished and ongoing data sets recommended by respondees were included as well. Despite the considerable effort, some experts were undoubtedly not contacted and it is likely that some number of references meeting our criteria were not discovered.

Subsequently, all literature contained in the bibliography but not presently in the EPA - Corvallis wetlands library was obtained. Study locations were plotted on state maps (Appendix B) using a geographic information system at the Lab (ArcInfo GIS), quantitative data were extracted and compiled for chapter tables, the "Methods" sections of papers were reviewed, and the narrative descriptions presented in the following chapters were prepared. Quality of individual data sets or their locations on the maps could not be checked or assured.

In addition, various national databases exist that frequently contain wetland community data. Data from sites associated with these databases were obtained and/or the site locations were plotted on the digital maps. These include:

- LTER network (all areas plotted; Long Term Environmental Research sites sponsored by the National Science Foundation);
- Christmas Bird Count database (all areas plotted); from Cornell Laboratory of Ornithology;
- Breeding Bird Survey database (all areas plotted); from U.S. Fish and Wildlife Service;
- Breeding Bird Census database (only wetland areas plotted); from Cornell Laboratory of Ornithology;
- Waterfowl Surveys (Migrating, Wintering, Spring Waterfowl Surveys, Summer Brood Count/ Breeding Ground Surveys); from Waterfowl Flyway Technical Representatives in each state;
- International Shorebird Survey (all inland wetlands); from Manomet Bird Observatory, Manomet, Massachusetts.

In addition, several data sets exist that may include relevant wetlands biological data, but with the limited effort of this project, such data could not be easily compiled or separated from non-wetland data. Examples of these include:

- wetland boundary determinations by consultants and agencies (a vast source of botanical data);

- measured data collected in support of HEP analyses by numerous consultants and agencies;
- river basin reports of government water quality monitoring programs (a source of fish and invertebrate data, if "wetland" stations could be separated from others);
- monthly bird counts of the National Wildlife Refuges;
- data from the Nest Card and Colonial Waterbird databases of the Cornell Laboratory of Ornithology;
- private notes of birders, botanists, and other naturalists.

Wetland data not included because of their failure to meet one or more of our criteria included the following:

- National Contaminant Biomonitoring Program data of the U.S. Fish and Wildlife Service (focuses on bioaccumulation and generally does not include measurement of community-level variables);
- Inventories of wetland threatened/ endangered species (not measurement of community-level variables);
- Inventories of wetland acreage and distribution (not measurement of community-level variables).
- Databases of The Nature Conservancy and state heritage programs (field data often not quantitative)

2.0 APPROACHES FOR PROTECTING WETLAND QUALITY

2.1 REGULATORY BACKGROUND

Statutes to reduce the impacts of the disposal of dredged and fill material in wetlands (e.g., Section 404 of the Clean Water Act) do not directly address impacts to wetlands from drainage, vegetation removal, and nonpoint-source discharges. However, other provisions of the Clean Water Act, if applied more vigorously to wetlands, have the potential to significantly reduce these impacts (USEPA 1989b). Moreover, interest in restoring degraded lands, including wetlands, appears to be growing, and because not all degraded areas can be restored immediately, priorities based partly on the existing degree of degradation must be developed. Questions arise, then, as to how best to measure, protect, and restore the quality of wetlands.

Many activities and discharges of pollutants into lakes and streams are regulated by State and Federal agencies. For example, under the State water quality certification authority of Section 401 of the Clean Water Act, States may grant, deny, or condition Federal permits or licenses that authorize a wetland alteration within that state. States are also mandated to develop and adopt water quality standards, as provided in Section 303 of the Clean Water Act, and all have done so. These standards must be applied to all waters of a State. The standards are the basis upon which States review permits to determine whether a proposed activity will meet a "use" that has been designated by the State for a particular water. Federal agencies reviewing applications for wetland alteration must comply with State decisions rendered under Section 401.

EPA, in its Water Quality Standards Program, requires State programs to include five components, two of which are the focus of this chapter:

- o Designating Uses
- o Applying Water Quality Criteria

The following sections of this report define these and describe optional approaches for States to consider as they address the future application of water quality standards to wetlands.

2.2 USE DESIGNATION AND CLASSIFICATION

"Designated uses" are uses or goals--such as public water supply, propagation of fish and wildlife, and recreation--that may be specified for each water body or wetland, whether or not they are currently being attained. Because of the high biological productivity of many wetlands, fish and wildlife uses are often emphasized, and the designated use category of "fishable/swimmable" that already covers other surface waters in most State programs can, as a first step, be administratively extended to cover wetlands. Use-designations may reflect either an acceptable current use of a wetland, or particularly in the case of restoration programs, a desired or attainable future use. They may be described in either general (e.g., "wetlands in Basin A should sustain commercial fishery production") or specific terms (e.g., "wetlands in Basin A should support a Fish Index of at least 3.5"). Multiple uses may occur or be designated within a single wetland or wetland type, and criteria appropriate for protecting one use may differ from criteria appropriate for protecting another.

Wetlands also typically have "uses" not commonly designated in State programs concerned with other surface waters. Examples include floodwater storage, groundwater recharge, shoreline stabilization, and wastewater purification. These uses (commonly termed "functions" by wetland managers) have been recognized, for example, in Section 404(b)(1) Guidelines of the Clean Water Act. Currently, few State water quality or wetland management programs have promulgated explicit procedures or standards for designating and protecting these uses.

Specifying a goal or "use" is not the only way for a State to protect the quality of its wetlands. Under the Antidegradation provisions of the Clean Water Act (40 CFR 131.12(a)(3)), States can simply declare all wetlands, all wetlands of a certain type, or particular wetlands to be "outstanding national resource waters." If the States' antidegradation policies are at least as protective as EPA's, this generally affords these wetlands the highest degree of protection because no degradation is allowed, except for "short term changes that have no long-term consequences" (USEPA 1989b). Other approaches, both regulatory and non-regulatory, might also be used for protecting wetland quality, including (but certainly not limited to) EPA's Advance Identification initiatives, nonpoint source management plans, water allocation negotiations, State Wetland Conservation Plans, State Conservation of Outdoor Recreation Plans (SCORP's), emission control programs, and others.

An obvious first step for any approach to regulating wetland water quality is to determine the general distribution and location of wetlands. The most comprehensive source of such information is the series of quadrangle-based wetland maps available for most of the United States from the U.S. Fish and Wildlife Service (obtained by phoning 1-800-USA-MAPS). These maps classify wetlands into a number of categories not necessarily related to their functions or uses. Although field-checking is often required to determine if wetlands on these maps (and possibly some that are not) are subject to regulation under Section 404, such intensive verification is not required for States to include wetlands under their definition of State waters.

As noted previously, existing use designations for State waters may be extended to wetlands. However, the inherent diversity of wetlands is best protected by developing different sub-categories of a use to different wetland types, at which point wetlands would be classified further. Designated uses, and criteria for protecting these uses, can be assigned as described below to each wetland, to landscape units of wetlands, or to each wetland type. To achieve this, two major options are presented: (a) Strategic Setting, and (b) Probable Functions.

The **Strategic Setting** option involves assigning designated uses to wetlands based on their landscape position, relative to connected waters or adjoining lands which potentially benefit from uses or services the wetlands provide. A rudimentary application of this approach would involve assigning to wetlands the same "use" currently designated for all waters into which they flow. Some technical consideration should be given regarding the nature of the hydrologic connection that exists between the wetlands and the receiving waters; even wetlands that lack surface water outlets are sometimes intimately connected to other waters via subsurface flow. A distance criterion may also be appropriate, because beyond some distance, the contribution of wetlands to certain functions of receiving water uses may, even on a cumulative basis, be indetectable. To reduce subjectivity, some simple models (e.g., Phillips 1989) might be used to assign technically-derived distance coefficients for use in different river basin types.

A conceptually similar but somewhat more descriptive option for assigning designated uses to wetlands can be used to supplement the above, and to prioritize wetlands for more detailed scrutiny in the use-designation process. Under this option, the designation of uses for a particular wetland is based on the presence of nearby cultural features that would be expected to benefit from functions typically performed by most wetlands. For example, a wetland might be assigned a use-designation of "water purification" if it is located a reasonable distance upstream from a cultural feature that relies on high-quality water, e.g., a public drinking water intake. In many but not all cases, such receiving waters may already have drinking water as a major use designation. Other cultural features that may benefit from typical wetland functions might include the following (these are only a few examples):

- sole source aquifers
- waters with known fish kill or eutrophication problems
- other waters believed to be in violation of water quality criteria

- waters in periodic need of dredging
- areas intensively managed or protected for their ecological resources
- floodplains containing economically significant development

In operation, this approach would begin by reviewing and, if appropriate, expanding upon the above list. Such a list might then be circulated for public input and perhaps narrowed to include just those for which geographic data are available. Then, these cultural features are mapped and their drainage areas or other functionally connected areas are delineated. Finally, a distance criterion is specified, wherein all wetlands containing, located upstream from, or otherwise functionally influenced by the listed feature (and within the specified distance) are considered to be strategically situated. That is, they are positioned so as to individually or collectively deliver or support the particular designated use. If no cultural features of concern are positioned within an appropriate distance from a wetland, the wetland may be assigned a general designated use that has been assumed for all wetlands statewide (perhaps "aquatic life") unless a use attainability analysis provides evidence to the contrary.

Once the existing geographic data have been assembled, the Strategic Setting option can be applied rapidly to large areas (river basins or entire states), because it mainly designates uses according to watersheds or drainage areas, rather than requiring wetlands to be evaluated individually. However, it makes no evaluation as to whether uses that are reputed to be provided by wetlands generally are actually provided by a **particular** wetland; it only evaluates whether a wetland is positioned so that such uses, if performed, will have an important recipient or user.

The approach does not require that all wetlands be mapped. Once the strategic watersheds have been identified, the responsibility for determining the locations of wetlands that are affected might be assigned to permit applicants. Managers must also keep in mind that downstream cultural features, and thus the strategic status of a wetland, can change with time. Accordingly, an overall designation of **attainable** uses should always be provided.

A second option, the **Probable Function** option, generally involves designating a use or uses to individual wetlands, wetland landscapes, or wetland types based on (a) direct measurement of the use, i.e., wetland function, or, (b) structural indicators of the use. Ideally, the uses or functions are measured directly in each wetland and their verification becomes the basis for establishing that designated use in each wetland. However, such individual verification of uses is seldom feasible without significant time and funds. Even then, uses verified to exist in wetlands under one set of annual climatic conditions may not exist in subsequent years under different climatic conditions.

An alternative is to employ structural indicators of wetland function--such as degree of channel meandering, watershed position, and connectedness--to modify more general descriptors provided by existing classification schemes such as the Cowardin et al. (1979) classification scheme. The literature on such structural indicators of wetland function has been summarized by Adamus and Stockwell (1983) and Adamus et al. (1990). Cursory evaluation of these structural indicators can be accomplished largely by reviewing airphotos and topographic maps depicting the wetland and its associated landscape, with perhaps a single brief site visit.

At the simplest level, the classification scheme of the U.S. Fish and Wildlife Service (Cowardin et al. 1979) might be used as the sole structural indicator of wetland function. Uses that are expected to be typically attainable for each wetland map category (e.g., riverine emergent, palustrine emergent, palustrine forested) are described, without regard for where on the landscape the wetland exists. Such an approach can be implemented quickly due to the availability of wetland maps, but some uncertainty exists as to whether map-based classification schemes designed for more general purposes are sufficiently sensitive to the wide variability in degree of function among wetland types.

2.3 DEVELOPING PRIORITIZED USE SUB-CATEGORIES

The description of the above two options (Strategic Setting and Probable Function) has focused on ways these options might be employed for designating uses of wetlands. Additionally, in some cases they might be used for (a) establishing sub-categories of use, and/or (b) establishing criteria that describe the conditions necessary to protect particular uses. This section focuses on (a), and the following section (2.4) addresses (b). Although most States establish standards for water bodies simply according to the uses they are designated to support, other States have established a hierarchy of uses with the higher uses denoting higher water quality. The former situation was described above, and the latter is described in this section.

Applying the Strategic Setting option to establish a sub-category of use might involve designating not only a general use such as "drinking water" to wetlands upstream of a drinking water intake, but also assigning a sub-category of use entitled "high quality for drinking water protection."

In another case, if the Probable Function option is used, wetlands which appear (for example) to support aquatic life might be assigned to a sub-category of use entitled "high quality wetland for shorebirds" based on direct functional measurement, Cowardin class, or structural indicators. Direct functional measurement could involve regional biosurveys and use of community-level indices to establish sub-categories descriptive of wetland quality, as some States have used for non-wetland surface waters. If, instead, structural indicators were used as the basis for establishing higher sub-categories within the "aquatic life" use, this could involve defining the "best" wetlands for this use in terms of their seasonal hydrology, vegetation, soils, landscape position, and other factors.

As part of EPA's "Advance Identification" program, some EPA regional offices, localities, and states are identifying or rating functions and uses of hundreds of individual wetlands using structural indicators or, rarely, direct functional measurement. Some States (e.g., Florida, Swihart et al. 1986) have similarly ranked wetlands as part of efforts to (a) designate "Outstanding Natural Resource Waters" under state water quality laws, or (b) designate wetlands of exceptional importance to waterfowl under State wetland conservation/recreation plans and the North American Waterfowl Management Plan. In these cases, existing evaluations might be used as one data source for considering appropriate sub-categories of designated use.

Regardless of whether use sub-categories are identified by direct measurement (e.g., biosurveys) or through use of structural indicators, considerable effort and time is required. Three options for making the task more tractable are available and are described as follows.

One option is to measure the functions (uses) in a limited set of "reference wetlands," which might be either a randomly selected set of regional wetlands, or a set of wetlands selected because they are believed to represent the least disturbed conditions. Once the reference wetlands are chosen, measurements of their structure and/or function (e.g., diversity, biogeochemical cycling rates, hydrologic transfer rates, community composition) might be used for defining the highest sub-category of protection. Such regional efforts would involve four steps:

1. "Reference wetlands" are chosen.
2. Functional (use) data are collected from these and compiled.
3. Spatial and temporal variability in the data, within and among reference wetlands, is compiled.
4. Use-subcategories are developed or modified.

If reference wetlands are selected using the random approach (e.g., Abbruzzese et al. 1988), measurements of these may not reflect "best attainable" levels of function, because in some regions, a majority of wetlands may not be functioning at desired levels, due to landscape-scale impacts. Conversely, if the "least disturbed" selection approach is used, some of the selected wetlands may not be providing some functions at levels desired by some segments of the public; for example, greater benefits to some components of aquatic life may be provided by wetlands that are actively managed rather than undisturbed. Agencies might desire that certain species or processes be the focus of protection in a particular wetland or watershed, and these might be dependent on continuation of existing management practices. Thus, if definition of "best attainable" functional condition is the desired objective, data from reference wetlands selected by either approach would serve only as a starting point. The reference wetland data would eventually need to be evaluated from the perspective of resource goals (as described above under Use Designation). Attainable condition could also be evaluated using data from from similar wetlands in less-disturbed adjoining regions, and/or from analysis of pre-settlement conditions.

To begin the selection of reference wetlands using the "least-disturbed" approach, attention would be given to identifying wetlands having superficial characteristics such as the following:

- wetland arose naturally and at a considerable time in the past, rather than being recently constructed;
- surrounding watershed, particularly within 500 feet of the wetland transition with upland, is largely undeveloped;
- water levels fluctuate naturally, not being affected by diversions, dams, or nearby wells;
- wetland has not been recently used for silviculture, grazing, or other human uses that potentially impact vegetation and/or water quality and quantity.

Frequently, when defining the "least disturbed" condition (either for an individual wetland or the regional wetland resource), the objective is to maintain the use within an "envelope" of expected temporal variability at a site or within a region. Although quantifying this can be a challenge, in some wetland types, recent developments in methods such as seed bank analysis (e.g., Poiani and Johnson 1989), tree ring analysis (Bowers et al. 1985, Hupp and Morris 1990, Sigafoos 1964), as well as sediment core analysis of pollen (palynological analysis)(Agbeti and Dickman 1989, Battarbee and Charles 1987) and sediment deposition (with measurement of lead 210 and/or cesium 137, see Bloesch and Evans 1982, Ritchie and McHenry 1985) can be used to identify the extent of hydrologic and botanical variability that has existed in both recent and distant historic times.

To select reference wetlands using the "random selection" approach, a statistical sample of all wetlands in each general category (e.g., riverine emergent, palustrine emergent, palustrine forested) is visited and probable functions of each wetland are assessed using structural indicators (as organized in any of several rapid methods for wetland evaluation, see Kusler and Riexinger (1986) for examples) or direct measurement of function (e.g., biological surveys). As enough wetlands are sampled to overcome variability within a broad class, generalities about the class in a particular region may begin to emerge. This approach was demonstrated by Schiefele and Mulamoottil (1988), who used structural indicators, and by Ohio EPA (1987), who measured functions (aquatic life values) directly, in non-wetland surface waters. These characterizations of the functions (uses) of wetlands of a general type then could be used for assigning distinctive levels or types of protection to specific wetland types. This approach also has the advantage that potential biases in selecting "least-disturbed" wetlands are avoided. By incorporating statistically representative data that will be collected beginning in the mid-1990's by EPA's Environmental Monitoring and Assessment Program (EMAP), the attractiveness of this approach may grow in future years.

Experiences of the EPA Wetlands Research Team suggest that, if a subset of a population of wetlands is to be visited and evaluated, the subset should contain 10 to 20 times as many wetlands than will actually be sampled, because denial of access is a common problem. Finding this amount of suitable wetlands when wetlands normally comprise less than 10 percent of the landscape can be a daunting task. In states with digitized (GIS-based) wetland maps, however, it can be much simpler.

If a decision is made to stratify the set of sampled wetlands, the degree of detail associated with the description may be important. For example, a stratification based only on broad general map categories such as "palustrine" and "lacustrine" wetlands would result in much higher variability than one in which the definition of "types" is based on structural features that are expected to best relate to the intended use or function. More samples of reference wetlands would be required to adequately define "typical" uses and functions. If, instead, types of wetlands were defined by a larger set of structural indicators (e.g., landscape position, channel meandering, soil type), variability among wetlands within types would be less and less monitoring would be required to characterize the typical uses of each type. For some purposes, the host of diverse structural indicators could be synthesized into fewer "types" by recognizing categories defined by expected biogeochemical forcing functions, e.g., wetlands dominated by flowing water vs. wetlands dominated by wave energy vs. wetlands dominated by ground water influx. However, without costly field-checking, wetlands cannot be reliably classified across entire regions according to such a scheme. Its application poses several operational problems with definitions as well.

A third option for making the Probable Function component more tractable is to focus evaluation at a landscape level, assessing function directly or measuring its structural indicators at a coarse scale rather than wetland-by-wetland. Regional information on structural indicators of wetland function used in such an approach might include soil types, runoff, and landform, as depicted on existing maps and geographic databases which summarize these. Although such data are seldom collected and compiled at a similar scale and level of resolution, for planning-level estimates they might be combined to yield qualitative estimates of the cumulative contribution of wetlands to landscape function in a region (Abbruzzese et al. In Press). Once the relevent data layers have been identified, acquired, and assembled, the categorization of similar landscape units and designation of their uses may proceed quite rapidly.

Regardless of which option is used for reducing the data collection effort of the Probable Function component, data interpretation will remain an important concern. Specifically, considerable subjectivity may surround decisions as to what numeric thresholds in the data should define particular sub-categories of use. Accordingly, the definition of use sub-categories should involve both public and technical inputs. Systematic procedures are available for helping reduce complexity and subjectivity in data interpretation (e.g., Krebs 1989). These include statistical approaches, e.g., systematically clustering data in groups, identifying quartiles or distributional nodes in the data, "break points" in cumulative frequency curves, and similar procedures. The eventual result is a sub-categorization of uses according to the degree to which they should be satisfied. For example, a State might wish to define "Class A" wetlands as:

o All bogs that contain greater than 80 % of the bog-dependent amphibians found in bogs of the same size in the region, OR,

o All wetlands that contain greater than 4 vegetation strata," OR,

o All herbaceous floodplains with a net annual productivity of greater than 2000 grams of carbon per square meter per year."

For a less pristine (e.g., "Class B") sub-category, the above figures might be relaxed to 70%, 3, and 1000 per square meter, respectively. Again, these specific levels are only illustrative, and would need to be derived from a biosurvey of wetlands in each region.

For greatest replicability, sub-categories of use describe the desired or actual condition of biota that can be directly measured, rather than only describing the sub-category as "degraded", "pristine", and similar qualitative terms. Biological descriptions contained in the descriptions are related directly to management goals. Descriptions of use sub-categories might include lists or ratios of organisms that characteristically dominate altered and unaltered wetlands, or organisms physiologically tolerant or intolerant of a particular type of stressor, if such information is regionally available for the particular wetland type. Descriptions of use sub-categories that include multiple trophic levels (e.g., algae, fish, birds) are more difficult to develop but may improve reliability and provide flexibility in applying assessment techniques. Descriptions of use sub-categories should also footnote particular assessment protocols (perhaps including methods for determining the requisite number and distribution of samples) used to develop the criteria and to be used to document compliance or non-compliance or to refine use sub-category descriptions.

2.4 NARRATIVE CRITERIA TO PROTECT WETLAND DESIGNATED USES

Once uses are designated, narrative or numeric criteria must be established to protect each use. EPA has indicated that, by September 30, 1993, States and qualified Indian Tribes shall apply standards to wetlands that incorporate, among others, designated uses, aesthetic narrative criteria (e.g., "free from..."), and narrative biological criteria, as well as appropriate numeric criteria. As States desired to become more protective of wetlands, these requirements are to be based on existing information. However, new information will be needed to eventually refine the standards.

At the simplest level, a State might define its narrative criteria is to state that no activity be permitted that results in a net loss of wetland acreage. This is because wetland extent is the most fundamental measure of wetland function. Wetland extent is best evaluated on a landscape or regional scale, as it is at this level that wetlands may provide the most significant benefits to aquatic life and wildlife. For example, the cumulative acreage of wetlands in a region, and the specific combinations and juxtapositioning of wetland types, can mean more to highly mobile waterfowl than does the contamination status of a particular wetland. This is because the daily and annual movements of many animals encompass several wetlands.

If wetland extent is to be used to define desirable sub-categories of use, it may be necessary to initially define "reference conditions" at a landscape level. For wetland extent, this may mean determining, from existing wetland maps and airphotos, mean densities of wetlands (acres per square mile), perhaps of various types and in various landscape contexts within a region. This metric could be determined just for "least-disturbed" landscapes, or for all landscape units in a region. Alternatively, an appropriate series of archival airphotos could be interpreted to yield information on historical wetland density (e.g., acreage of wetlands per square mile).

At a somewhat more detailed level, narrative criteria could specify that wetlands shall not be changed from one Cowardin type to another. This would be based on an assumption that maintaining a particular Cowardin type maintains the designated uses. However, wetland science indicates that considerable degradation of a wetland's functions (uses) can occur without being manifested as a change in Cowardin type.

In contrast, if structural indicators are used (as described above in descriptions of the Probable Function option), uses might be somewhat better-protected. For example, narrative criteria might specify that no activity be permitted that decreases a wetland's probability rating for "Aquatic Diversity," as indicated by an accepted, structure-based wetland evaluation procedure. However, wetland science also indicates that structural indicators of wetland function are not always reliable; wetland function can be considerably degraded without obvious signs of structural change. Also, wetland evaluation methods have not been designed to distinguish which of the structural features they employ are **determinants** of wetland function (and thus useful in criteria development) vs. **indicators** (mere correlates) of wetland function. In either case, these structural factors may not currently be subject to legal regulation and/or may not normally be altered by development. Also, protection of the structural integrity of wetlands in one county or state may or may not guarantee against degradation of the associated use that results from stresses that occur beyond jurisdictional boundaries.

2.5 NUMERIC CRITERIA TO PROTECT WETLAND DESIGNATED USES

Where the predominant stress to a wetland is one that is commonly regulated in other surface waters (i.e., Contaminant Toxicity, Enrichment/Eutrophication, Organic Loading/Dissolved Oxygen, Acidification, Salinization, Turbidity, Thermal Alteration), the existing aquatic life criteria (USEPA 1986) that are commonly applied may also be applied to wetlands. In addition, it can be assumed that all human health criteria for other surface waters apply to wetlands. In the longer term, regionally-based numeric criteria might be developed for wetlands by measuring functions/uses directly, as opposed to narrative criteria which would rely mostly on Cowardin type or structural indicators.

In the case of "aquatic life" uses of wetlands, this could involve eventually developing biocriteria for wetlands. If regional biosurveys are not immediately feasible as a basis for developing biocriteria, existing field data sets interpreted with great caution might sometimes be used. For example, existing data may be sufficient to indicate the approximate number and proportions of particular species expected to occur in a particular wetland type, and this knowledge could be used to help establish biocriteria protective of that general use. However, any such criteria derived from the literature must consider the likely non-representativeness of the data and potential biases arising from species-area effects and variable levels of effort.

One concern that arises is that existing narrative and numeric criteria are inadequate to address some impacts that critically affect wetland function and use, such as hydrologic and physical alteration. Of the stressors listed in Table 5, the physical impacts of Dehydration, Inundation, Vegetation Removal, Sedimentation, Shade, and Habitat Fragmentation in particular are currently seldom addressed by water quality protection programs. The impacts to wetland uses from these stressors are likely to often exceed impairments to use resulting from chemical contamination.

Existing information is sufficient to develop narrative criteria to address these stressors. However, there are probably insufficient data to promulgate numeric criteria at present, except perhaps site-specifically for a few well-studied wetlands. Future, long-term development of numeric criteria for protecting wetland uses against physical alteration may require new research protocols, such as expanded use of field mesocosm and whole-wetland or whole-watershed manipulations. Laboratory testing--the typical approach used for contaminants--would be less appropriate due to the scales involved and the complexity of interactions. Also, if numeric criteria were to be developed for physical stressors, the criteria would need to be keyed in to specific uses, because (for example) the amount of sedimentation that is detrimental in a wetland to some uses might be beneficial to others.

In the long term, however, some chemical criteria may need to be re-evaluated site-specifically because of the unusual conditions encountered in some wetland types. Undisturbed wetlands sometimes have lower pH and dissolved oxygen; higher organic carbon, humic acids, temperature, ammonia, and sulfide; extreme reducing conditions; more potential for photodegradation, biodegradation, chelation and organic complexation than do surface waters generally and laboratory waters specifically. Under certain circumstances and for some contaminants, these conditions can profoundly affect the rate and direction of contaminant mobility in wetlands, as well as the bioavailability and toxicity of these contaminants (e.g., Winner 1984). Moreover, the spatial and temporal variability of these conditions is believed to be much greater in wetlands than in non-wetland waters, due to their shallow depths, prevalence of vegetation, and closer dependence on hydrologic forces.

In applying existing numeric criteria to wetlands, careful consideration is most appropriate when (a) the laboratory water used to develop existing criteria differs significantly from extremes found in a particular wetland type, and/or (b) the types of organisms in the region differ significantly in their physiologic responses or propensity for bioaccumulation from those used in testing. Thus, in cases where verification of the applicability of existing numeric criteria to a particular wetland or wetland type is essential, a four-phase procedure might be used, involving (a) review of data describing laboratory water chemistry and indicator species used previously to develop the criteria, (b) analysis of water samples or sediments from the subject wetland or wetland type to determine if they are significantly dissimilar from laboratory water used in testing, (c) biological survey of species inhabiting the subject wetland or wetland type to determine if they are significantly dissimilar (in terms of physiologic responses or bioaccumulation potential) from species used in the laboratory bioassays, and (d) rerunning the toxicity tests, if necessary as indicated from the results of (a)-(c).

If re-running of toxicity tests is desired and feasible, ideally, a tiered testing scheme is used (Kimerle et al. 1978, Ongley et al. 1988). Traditionally, risk is assessed and numeric criteria are developed by incorporating a hierarchy of toxicity tests of increasing complexity, chemical data, and expected exposure regimes (La Point and Perry 1989). This involves use of a combination of single-species laboratory assessments of acute toxicity, field microcosms, field mesocosms, and modeling of toxicity, transport, and fate based on chemical structure and other factors (Matthews et al. 1982). This could be done using bioassays featuring indigenous wetland organisms (e.g., Fremling and Mauk 1980, Lee et al. 1987), and/or by manipulating experimentally-confined wetlands to determine biotic responses (e.g., Carpenter and Chaney 1983, Hurlburt et al. 1972, Richardson et al. 1983). EPA protocols specify that establishment of an acute value for a freshwater criterion be based on a minimum of eight different taxonomic families, including a freshwater alga or vascular plant, a planktonic crustacean, a benthic crustacean, an insect, a nonarthropod nonchordate, another insect or a new phylum, a salmonid, another fish, and another chordate. Partly because such testing has been carried out chemical-by-chemical, the development of criteria has typically been a lengthy process and efforts have only recently begun to better define chemical interactions, through use of whole-effluent toxicity testing and other approaches. However, the criteria which result are regarded as simple to apply and interpret, thus allowing regulation of an effluent to be undertaken incrementally through licences and permits. Consideration of the need for re-testing might focus initially on substances whose criteria were based on testing of the fewest species, because the probability is less that these would include a sufficient number of wetland species to fulfill the EPA requirement for bioassay of at least eight taxonomic families.

Given the large number of wetlands potentially exposed to contaminants, the costs associated with such site-specific testing might be justified only where existing biochemical data had indicated that a particular wetland was significantly different from biochemical test conditions. Because there is seldom enough available data on background biochemical conditions of large numbers of wetlands to indicate that they differ significantly from test conditions, it may be necessary to either (a) use exposure indicators (e.g., proximity to hazardous waste sites) or administrative needs (e.g., permit applications) to select wetlands for site-specific testing or modification of criteria, or (b) statistically select a representative series of wetlands, stratified by their probable, naturally-occurring biochemical type, that are suspected to deviate the most from test conditions, and then confirm their biochemical categorization with field measurements and re-test their biota. Any resulting modifications to existing criteria would be applied to the entire regional population of wetlands of that biochemical type.

Future applications of numeric criteria to wetlands could include performance standards, impact standards, or both (Courtemanch et al. 1989). Performance standards are characterized by a focus on each pollutant, and are commonly expressed as "end-of-the-pipe" or "receiving water" desired concentrations or loadings. These are often specified in terms of allowable magnitude, duration of exposure, and frequency, and are designed to protect aquatic life both from risks due to bioaccumulation and from acute and chronic toxicity. In contrast, impact standards require that a certain result be achieved. They are typically specified in terms of biocriteria for ambient waters or sediments, e.g., desired species composition and richness, as described in the earlier.

As States begin to protect and restore wetlands through a biocriteria approach, a question arises as to which features, processes, or organisms best indicate the ecological "health" of the wetland resource, or are desirable due to convenience of monitoring or other reasons. Because EPA's national goal for wetland protection is "no net loss in acreage or function," it may be desirable to additionally examine the community structure and processes within wetlands, to establish criteria for biological function and to monitor attainment of the functional quality goal. This can be done using the approach described above, i.e., identifying reference conditions, compiling data, analyzing variability, and ultimately establishing use-designation criteria or setting restoration priorities--either through field surveys or professional consensus. However, in doing so, one faces the questions:

o "What are the best indicators of wetland biological function?"
o "How to monitor wetland biological function?"

These are the subjects of the next chapter.

3.0 GENERAL GUIDELINES FOR WETLANDS BIOLOGICAL CHARACTERIZATION

Wetlands pose unusual challenges for monitoring programs. Because wetlands, as transitional environments, are located between uplands and deepwater areas, their biota exhibits extreme spatial variability, triggered by very slight changes in elevation. Temporal variability is also great, because the shallowness of any surface water results in its being highly influenced by slight, fleeting changes in precipitation, evaporation, or infiltration. Only a minority of all wetlands in the United States have permanent surface water (Shaw and Fredine 1956), so sampling techniques developed for other surface waters are not always applicable. The extreme spatial and temporal variability often requires that large numbers of samples be collected if the wetland community is to be properly characterized.

Such extensive sampling is made difficult, however, by potentially severe problems of access. Physically, access to many wetlands is hindered by water too shallow for rapid boat access, soil too fluid for rapid foot or vehicular access, and vegetation canopies too dense for easy aerial or airboat access. Access to many wetlands is also seriously hindered by the widespread (and sometimes misguided) public perception that wetlands, in contrast to other waters regulated by the Clean Water Act, are exclusively private land. Landowner awareness of the potential for regulation has led to commonplace denial of requests for access to wetlands during other EPA projects. Proportionally few wetlands are publicly owned, and these are not necessarily representative of the total wetlands population. These factors all combine to potentially increase the costs of an effective wetland monitoring program, and pose significant demands for study design and logistical planning.

Despite these difficulties, the need for more vigorous wetland sampling efforts is compelling. Because most wetlands are located in a topographically low, depositional environment and have long hydraulic detention times, they accumulate contaminants from a wide area. At the same time, undisturbed wetlands are characterized by exceptional biological productivity, suggesting a greater need for more extensive monitoring of wetlands. However, wetlands seldom are monitored, so much remains to be learned about the extent to which contamination and other stressors have altered their condition.

3.1 WHAT TO MONITOR

Monitoring of **multiple indicators**--having both short and long lifespans, and both localized and broad home ranges--is preferable to monitoring a few because indicators differ in their sensitivity to different types of stress in different types of wetlands, and in their temporal and spatial occurrence. By monitoring both short- and long-lived taxa, the effects both of stressors that occur briefly (e.g., herbicides) and of those that occur over longer time periods (e.g., bioaccumulation of metals) can be detected. By monitoring both resident and wide-ranging/ migrant species, the cumulative landscape-level impacts that may not be detectable on a local scale may become apparent. Ideally, **monitoring of a wetland should encompass as long a time period, as many indicators, and as many microhabitats within the wetland as possible, given available resources.** However, the need to make choices is inevitable.

Another choice concerns the which level of ecological hierarchy should be measured--e.g., physiology of individuals, demographics of a population, structure of a community, or processes of an entire ecosystem. Conclusions from one level cannot necessarily be extrapolated to another. As noted in Chapter 1, the scope of this report is limited primarily to the community level. A good discussion of factors affecting the choice of an appropriate hierarchical level in wetlands is presented by Farmer and Adams (1989).

Sometimes, the analysis of initial data collections can be used to target particularly sensitive groups or processes and identify optimal numbers of samples. Also, if life histories and ecological relationships are sufficiently well-understood in a particular area, monitoring could be limited to a few taxa known for their sensitivity to a particular stressor or their role as ecological "keystones." Keystone species include those which physically alter the landscape so profoundly that they create or destroy habitat for a much larger group of species over a wide area.

Examples of taxa that are considered to be keystones in particular regions or wetland types include:

- woodpeckers, which excavate cavities required by dozens of species;
- bees and other pollinating or seed-dispersing organisms, which control habitat structure through their major collective effects on vegetation;
- gopher tortoises and other burrowing species that create shelter critical to survival of many other animals;
- beaver, which create wetlands and temporarily destroy forest;
- muskrats, alligators, and some herbivorous birds, which through grazing and physical movement cause locally major increases in open water patchiness of wetlands.

Caution is necessary because it is seldom possible to validly infer trends in all species by monitoring only one or a very few "keystone" or "indicator" species. Thus, changes in community-level metrics usually give a clearer indication of "abnormal" biological stress than does the presence or absence of a single indicator species, regardless of its reputation as a keystone (Browder 1988, Cairns 1974, Couch 1982, Grigal 1972, Hellawell 1984, Karr 1987, Kelly and Harwell 1989, Landres et al. 1988).

In other aquatic systems, stable isotope techniques have been used to help identify keystone species, ecosystem components, or processes. In the case of vascular plants, attempts to identify the most sensitive species have also been made by measuring exposure of a host of species to a particular substance (e.g., a nutrient) and then monitoring the varying degrees to which the substance accumulates in tissue (e.g., Canfield et al. 1983), or alters germination and other physiological processes. Species which accumulate the substance and/or show the greatest physiological response the most are presumed to be likely to be affected if the substance increases.

To identify the most sensitive indicators, greater efforts could be made to comb the literature on experimental toxicology. However, although use of standardized conditions in most toxicity testing allows some degree of comparison among taxa regarding their relative sensitivities, the usefulness of laboratory toxicological data can be limited by the dissimilarity of test conditions and typical wetland conditions (e.g., altered toxicant mobility and toxicity due to increased organic carbon; interactions between hydroperiod effects and chemical toxicity--see Chapter 2.0).

Conceptual models (e.g., Patterson and Whillans 1984) or simulation models (e.g., Summers and McKellar 1981) of wetland ecosystems also could be applied to identify impact networks and thus, taxa that are likely to be most vulnerable to a particular stressor, and/or are potential keystones in ecosystem energy flow (Levins 1973). However, modeling approaches are also limited by lack of data on many wetland species and stressors (e.g., tolerance of wetland organisms to desiccation, burial).

Inevitably, the choice of what to monitor is governed by both policy and scientific considerations. The following criteria (derived from Hellawell 1984, Kelly and Harwell 1989, Landres et al. 1988, Schaeffer et al. 1988, and Temple, Barker, & Sloane 1989), may apply:

Decision factors related to policy implications:

- Unambiguous - The indicator is socially relevant and easily understood as an indicator of ecological integrity and/or health;
- Evaluative - The indicator is capable of evaluating the effectiveness of regulations, control, or management strategies;
- Cost-effective - The indicator is capable of giving a maximum amount of information for a minimum cost, and thus fiscally attractive;
- Accessible - The indicator is capable of being generated from accessible data sources;
- Anticipatory - The indicator is capable of providing a warning in time to avoid widespread or irreversible damage.

Decision factors related to scientific implications:

- Sensitivity - The indicator is responsive to the range of conditions likely to be encountered;
- Common - The indicator is sufficiently present in wetlands to be captured by reasonable sampling effort;
- Integrative - The indicator is capable of integrating effects over time and space;
- Standardized - The indicator is either broadly used and possessing standard methods, or capable of development of standard methods;
- Reliable - The indicator provides comparable results over a wide range of conditions;
- Predictive - The indicator provides a predictable response to a given stressor or set of stressors;
- Rigorous - The indicator is scientifically accurate, precise, explicit and capable of standard measurement and reporting protocols that are congruent with the data quality objectives.

The relative weights given each of these evaluation factors will vary depending on the programmatic context, i.e., for which of the following potential purposes the indicator is being used:

- Determining simply whether a wetland is changing, and in what direction;
- Assessing how aberrant is the community structure of a particular wetland, e.g., to set priorities for restoration or strategies for mitigation;
- Evaluating the success of management of a wetland, e.g., compliance with permits and mitigation plans;
- Pinpointing the source of degradation of a wetland;
- Evaluating overall program success of wetland quality protection efforts;
- Priority ranking of wetlands;
- Gaining an understanding of fundamental wetland processes and advancing the science.

As we examined the technical literature on the most commonly monitored taxonomic groups, we applied the unweighted criteria to the indicators in a non-systematic, qualitative manner. A resulting summary of the advantages and disadvantages of each taxonomic group is presented as Appendix A. As data become available, a more thorough analysis would consider, more specifically, the differences among taxa with regard to particular stressors in particular wetland types.

To date, there appears to be only one field study (Brooks et al. 1990) that has attempted to compare the relative sensitivity of major phyla (at the level of community structure) for indicating anthropogenic stress in inland wetlands. Experimental studies making such comparisons are also virtually non-existent. Future efforts to develop and compare indicators could focus on studies that circumstantially span a gradient of disturbed and undisturbed (but otherwise as similar as possible) wetlands of all types. They could compare all taxa, metrics and data reduction techniques, which, from a theoretical perspective and studies to date, show promise for use (e.g., Do vegetation similarity measures respond more sensitively to heavy metal pollution than does wetland invertebrate biomass?). Such future efforts to develop and compare metrics could emphasize comparisons under different types of temporal and spatial variability.

Given this situation, an alternative approach is to query wetland experts regarding their personal opinions of taxa and metrics that might be most useful for a given purpose. Some of these opinions have been published (Table 6). However, recommendations can be unintentionally colored by the expert's degree of experience with a particular taxon.

As resources allow, rigorous approaches to indicator evaluation might involve integrated laboratory and field dosing experiments, conducted in parallel with empirical field studies of a series of wetlands that are as similar as possible but are situationally exposed to various levels (i.e., a gradient) of the same stressor. This is proposed in EPA's implementation plan for wetland - water quality research (Adamus 1989).

3.2 TYPES OF MONITORING

Monitoring methods might be classified as **qualitative** and **quantitative**. Qualitative methods are generally faster, based largely on visual observation, require little or no sampling equipment, and are usually applied just along the edges of wetlands. Compared to measurement-based quantitative methods, qualitative methods are often less replicable and accurate.

One type of qualitative method used in wetland biological monitoring involves use of ground-level (or low-level) photography. This typically consists of establishing fixed stations at several points around or within a wetland and taking photographs at specified times. Stations may be surveyed in to known benchmarks to assure that they may be subsequently located with accuracy, or objects expected to be immobile over time (e.g., heavy metal stakes) may be included in each picture. Range poles can also be included in pictures to document scale. Photographs are often pieced together to form a panorama, and video cameras are being used increasingly to comprehensively document conditions. Photographs can subsequently be evaluated visually, primarily for major changes in woody vegetation. Time-lapse photography can be used in some settings to monitor wildlife use. Cameras tethered to balloons have also been used in emergent wetlands to record interspersion of open water areas with vegetation, and distribution of submersed macrophytes (e.g., Edwards and Brown 1960).

Table 6. Wetland Monitoring Indicators Suggested by Various Scientists.

Aust et al. (1988):

These authors studied silvicultural impacts to wetlands, and found that the most efficient indices of changes in ecological function (from helicopter logging, skidding, and herbiciding) were soil acidity, redox potential, oxygen concentration, temperature, soil mechanical resistance, sedimentation, and vegetation cover. These require short sampling periods, a minimum of laboratory work, and easily operable and maintainable equipment. Less complex to interpret were sedimentation, net primary productivity, plant N and P uptake, cellulose decomposition, and bird richness, diversity, and abundance. Most responsive to disturbance (i.e., showing significant differences across gradients or between treatments) were total N and P concentrations in soil water, soil acidity, redox potential, saturated hydraulic conductivity, temperature, soil mechanical resistance, sedimentation, net primary productivity, plant N and P uptake, and cellulose decomposition. Most integrative of ecological processes were soil redox potential, net primary productivity, plant N and P uptake, and cellulose decomposition rates.

Brooks et al. (1989) and Brooks and Hughes (1988):

For general monitoring of inland wetlands, the following monitoring parameters were suggested: hydrology, water quality, hydric soils, vegetation (richness, density, productivity, vertical stand structure, horizontal patchiness), macroinvertebrates, fish, amphibians, birds, mammals.

Brown et al. (1989):

They proposed the following (in approximate priority order) be monitored for EMAP (EPA's proposed Environmental Monitoring and Assessment Program for wetlands, in which a probability sample of 3000 wetlands (50-100 of each of about 13 types) nationwide would be visited once every 3-4 years, with perhaps more-frequent airphoto coverage):

1. Regional changes in the acreage, type diversity, and spatial patterns of wetlands.
2. Nutrients
3. Other pollutants in sediments
4. Hydroperiod
5. Vegetation (patterns, abundance, richness, composition)
6. Sediment and organic matter accretion
7. Waterbird abundance and species composition
8. Bioaccumulation
9. Macroinvertebrates (abundance, biomass, composition)
10. Leaf area, percent light transmittance, greenness
11. Microbial community structure
12. Bioassays and biomarker measurement

Florida DER (Schwarz et al. 1987):

For state-required monitoring of wooded and cattail-dominated wetlands receiving treated wastewater, the following parameters are measured: water quality, detention time, vegetation ("importance value" of dominant species), macroinvertebrates (Shannon diversity index), and fish (biomass ratio of rough fish to sport and forage fish).

Kadlec ([illegible]88):

Chemical inputs and outputs normalized to flow, vegetation biomass, sediment and organic matter accretion.

USEPA (1983):

For monitoring of wetlands receiving wastewater, the following parameters were listed: hydrology, nutrients, other dissolved substances, trace metals, refractory chemicals, sedimentation, vegetation (species composition, areal distribution, biomass, growth, production), detrital cycling (organic matter accretion), bioaccumulation, macroinvertebrates, fish (productivity, biomass, spawning success, bioassays, incidence of disease), wildlife communities (habitat structure, species richness, density, indicator species, incidence of disease).

USEPA (Sherman et al. 1989):

For comparison of multiple sets of constructed wetlands with reference wetlands in Florida, New England, and the Pacific Northwest, the following were measured: water depth, depth to water table, ambient nutrient concentrations, sediment chemistry, soil oxidation, morphometry and bank slope, vegetation (species composition, cover, natives vs. exotics).

Qualitative methods such as these can be used to develop maps of vegetation within wetlands, e.g., Farney and Bookhout (1982), Meeks (1969), and Morgan and Philipp (1986). Use of cover maps, aerial photos, and ground photos can be used to identify broad changes in plant composition, as well as providing permanent records. Suitable, existing, low-altitude color photographs often can be obtained from state offices of the Agricultural Stabilization and Conservation Service, from the U.S. Forest Service (forest pest management monitoring programs), and (near roads) from state highway departments, as well as other sources. Remote sensing has also been used under ideal circumstances to estimate soil saturation, primary productivity, and sedimentation (He[illegible]man 1982).

A second type of qualitative monitoring involves making visual, ground-level estimates simply of presence/absence of indicator species and physical conditions (e.g., Terrell and Perfetti 1989) and, in the case of vegetation, of percent cover. Vast numbers of such unpublished "species lists" are available from university botanical visits to wetlands, consultant reports, and other sources. While preferable to no data at all, these represent the "data rich - information poor" syndrome. However, some investigators go beyond a simple listing of species and visually estimate abundance in relative terms, e.g., rare, common, and this allows improved interpretation of data. Examples are reports by Dunn and Sharitz (1987), Ehrenfeld 1983, Kadlec and Hammer (1980), Nilsson and Keddy (1988), Taylor and Erman 1979, Wilcox (1986).

A third type of qualitative monitoring approach involves the use of "wetland evaluation" methods. Many such methods are available (e.g., see reviews by Adamus 1989, Kusler and Riexinger 1986, Lonard et al. 1981), but differ little in terms of their time requirements. Perhaps the most widely used are:

- Habitat Evaluation Procedures (HEP) of the U.S. Fish and Wildlife Service.
- Wetland Evaluation Technique (WET) developed by EPA and the Corps of Engineers (Adamus et al. 1987, Adamus et al. 1990).

Although these methods may benefit from or require a limited number of field measurements, they are predominantly qualitative. They do not directly measure biological communities, but rather, assume biological community structure or wetland function using information on habitat structure (Schroeder 1987). Most are applicable at the individual-site level (e.g., WET), while others (e.g., the "Synoptic Approach"--Abbruzzese et al. in press) operate at regional scales and require more cursory data inputs.

Quantitative methods are the focus of this report. Although many reports and books describe protocols for biological sampling of lakes and flowing waters, few have attempted to comprehensively describe or evaluate sampling modifications appropriate for the highly variable, transitional environments of wetlands. Some relevant information can be found in the following:

> Brooks 1989, Brooks and Hughes 1988, Erwin 1989, Fredrickson and Reid 1988a,b, Harris et al. 1984, Horner and Raedeke 1989, Murkin and Murkin 1989, Platts et al. 1987, USEPA 1986, Welcomme 1979, Woods 1985.

Other parts of EPA's Wetlands Research Program have developed protocols for wetland sampling. For example, at the EPA-Corvallis Laboratory, the Wetlands Team has developed protocols for monitoring created or restored (mitigation) wetlands (Sherman et al. 1989). The EPA-Duluth Laboratory has developed protocols for biological field-sampling of wetlands impacted by a variety of stressors. EPA is refining these and developing other protocols for support of its nationwide Environmental Monitoring and Assessment Program (EMAP). This report is not intended to substitute for these other protocols, but rather, includes them in discussions of a full range of methods available.

3.3 STUDY DESIGN

For detecting wetland ecological change and estimating its causes, a statistically powerful approach would involve sampling both before and after the expected change, in both exposed wetlands and in similar, unexposed wetlands (or in a large, random sample of similar wetlands with unknown exposure history). Selection of unexposed, or "reference" wetlands is discussed briefly in section 2.1. Statistical issues associated with wetland biomonitoring are discussed in greater detail by Simenstad et al. (1989).

Because of the high variability of wetland environments, sample collections should be replicated, both within and among wetlands, and within and among sampling times. One simple option for estimating the minimum effective number of samples or hours of effort involves plotting a curve. The "x" axis of the curve would describe the number of samples collected and the "y" axis would describe the community metric being measured (e.g., cumulative number of species), or its cumulative percent error or variance. Assuming a reasonably large number of samples have been initially collected, the point where the curve levels off might be considered to represent the minimum effective sampling effort. Statistical protocols are also available for estimating requisite number of samples in wetlands, given a desired detection level and initial information on sample variability (e.g., Downing and Anderson 1985, Eberhardt 1978, Jackson and Resh 1988, Resh and Price 1984).

There are several options for placement of sampling stations. In previous wetland studies, stations most often have been situated in one of the following ways:

- randomly;
- along transects (usually perpendicular to wetland gradient or flow and extending to the deepest part of the wetland, and sometimes intentionally aligned to intersect all habitat or topographic "types" within the wetland);
- at ecotones (spatial boundaries between major vegetation types, and open water and vegetation);
- in proportion to occurrence of habitat types (or hydroperiod classes) present within the wetland;
- at locations subjectively felt by the investigator to "represent" the wetland.

Seasonal timing of sampling is also important, and can be scheduled to coincide with (a) times at which organisms of concern are most likely to be at maximum numbers, (b) times when these organisms are most physiologically sensitive to a particular stressor, and (c) times at which concentration of, or organism exposure to, the stressor is greatest. From this, it is apparent that cost-effective wetland biomonitoring requires knowledge of (a) life history aspects of wetland organisms, (b) physiology and relative sensitivities to stressors of the component organisms, and (c) dynamics of physical and chemical factors that largely determine stressor availability. Most biological surveys of wetlands have been conducted during the growing season, and relatively little is known of exposure or community structure and function during stressful conditions of ice cover, severe anoxia, or drought. Time-of-day is also an important consideration, particularly when monitoring vertebrates. Unless diurnal behavior patterns are well-understood, or there is sufficient labor available to sample wetlands simultaneously, it may be desirable to alternate the order in which wetlands are visited, to avoid temporal bias.

The optimal seasonal timing from a biological perspective may not coincide with the best timing from a perspective of physical human access. Physical access into wetlands is notoriously difficult, and the more accessible edges of a wetland do not represent the biological conditions in a wetland generally. Although interior parts of wetlands may be more accessible during ice cover or drought, seldom are these the most biologically appropriate times for sampling. Previous investigators have used hip boots, canoes, inflatable rafts, airboats, helicopters, snowshoes (in summer, for distributing weight in peat bogs to prevent sinking), and scuba gear for dealing with problems posed by the semi-fluid substrate of many wetlands. For vegetation, remote sensors can be used for general coverage estimates. Low-altitude video can provide digital data directly, facilitating spatial analysis (pers. comm., M. Scott, U.S. Fish and Wildlife Service, Fort Collins, CO). Biomass of submersed aquatic macrophytes was measured electronically by Canfield et al. 1983, Duarte 1987, and Thomas et al. 1990.

3.4 DATA ANALYSIS AND INTERPRETATION

After addressing the question, "What should we measure?" the next logical question is "How do we express the data?" Thus, in developing and applying wetland biocriteria, the selection and interpretation of appropriate metrics is at least as important as the selection of appropriate taxa and sampling techniques. Questions related to wetland metric selection, such as the following, must inevitably be addressed if "data" are to be converted to "information:"

- Is abundance, biomass, or species richness a more sensitive indicator of wetland biological change?
- When are "guilds" an appropriate way to compile data?
- Do similarity indices and ordination procedures indicate stress from contaminants better than they show stress from hydroperiod alteration?
- When metrics describing ecosystem structure (such as the above) show that a wetland has changed, what can be inferred about the wetland's change in function?

Providing a detailed description of all possible techniques for analysis and interpretation of wetland data was considered beyond the scope of this report. Similarly, the validity and sensitivity of various metrics and procedures, as applied to the specific taxa and stressors described in later chapters, is not evaluated by this report. Such an evaluation, perhaps using the evaluation factors listed in section 3.1, would be extremely important in developing biocriteria for wetlands, but is not currently feasible due to lack of sufficient comparative data. Some of the more commonly used metrics and analysis procedures are shown in Tables 2, 3, and 4. Review and comparisons of performance of various indices in non-wetland ecosystems are given, for example, in Green and Vascotto 1978, Huhta 1979, Krebs 1989, Magurran 1988, Matthews et al. 1982, Polovino et al. 1983, Wolda 1981, Washington 1984, and others. For further information on statistical analysis of wetland community data the following references (among hundreds) might be consulted: Gauch 1982, Green and Vascotto 1978, Hill 1979, Isom 1986, Jongman et al. 1987, Ludwig and Reynolds 1988, Pielou 1984, and Wiegleb 1981.

Community-level metrics can also vary greatly in their sensitivity for detecting environmental stress. To optimize detection of ecologically degraded condition, **it is usually best to use several metrics or procedures in combination** (Schindler 1987), as is done by the "Index of Biotic Integrity" that was developed for other surface waters (Karr 1981).

For other surface waters, information compiled by Sheehan (1984) and others suggests that the approximate statistical sensitivity of community-level metrics/procedures to pollution has often been:

cluster/ordination > similarity > richness per unit area or effort> biomass/abundance
procedures indices and diversity indices

However, generalizations such as this contain a high degree of uncertainty. This is because of biases potentially arising from unknown (and perhaps inconsistent) dependence on a metric's or procedure's sensitivity to (a) statistical properties of the data set, (b) the particular combination of taxa contained in the data set (and associated life histories varying from sample to sample), (c) taxonomic level-of-identification, (d) wetland or community type, (e) spatial scale of measurement, (f) temporal scale of measurement (e.g., frequency of sampling, time elapsed since the stressor was maximal), (g) sampling equipment, level-of-effort, and techniques used. Thus, when only a few metrics and statistical procedures can be applied, results may be difficult to interpret. Unfortunately, few wetland studies have examined these potential biases. Also, of particular interest would be (a) the correlation of responses of metrics at several ecological levels, e.g., do metrics based on response at the organism level show the same response as those based on data from the population, community, and ecosystem levels? and (b) the correlation of responses of metrics to responses in ecosystem function (processes).

A single number from a metric, if used alone, sometimes provides little useful information. Often more instructive is the particular taxonomic composition that led to a particular summary metric value. Thus, where data on sensitivities and life histories of organisms are available, aggregating species-level monitoring data by functional groups ("guilds", see Table 4) of species can provide for more meaningful data interpretations. It can also reduce the statistical variability in data sets, thus reducing the number of requisite samples (pers. comm., Dr. James Karr, University of Virginia).

Moreover, shifts in taxonomic composition in response to contaminants frequently are likelier to occur than changes in total number of species or biomass (e.g., Ferrington and Crisp 1989). However, predicting which species will become dominant following a wetland disturbance is generally more difficult than predicting that species composition, overall richness, or biomass-abundance will change (Nilsson and Keddy 1988). In wetland macrophyte communities, richness is frequently correlated with biomass (Nilsson and Keddy 1988). This is not true in some communities of wetland fish (Tonn 1985).

All of the above metrics/procedures, except biomass/abundance, commonly employ species-level data. Such data are easily determined for taxa such as birds, but are much more difficult to acquire for microbial communities, which have large numbers of species, and for which comprehensive regional references on taxonomy are virtually non-existent. The need for species-level identifications for the determination of anthropogenic effects has been asserted by some studies and disputed by others; the need may depend on the factors listed above that pertain to metric biases, as well as on costs of making more-detailed identifications vs. costs of collecting a larger number of samples that are only identified at a general taxonomic level.

The utility of some metrics and procedures, as well as their sensitivity, may vary by wetland type. For example, metrics and procedures that depend on species-level data (richness, ordination, similarity indices) may be ineffective in describing the ecological condition of wetlands that characteristically have low species richness (e.g., breeding bird richness in salt marshes, fish richness in montane wetlands).

The metrics and procedures listed in this report represent only our current abilities to quantify wetland community structure. From the emerging discipline of "stress ecology" (e.g., Lugo 1978, Odum 1979, 1985), there may be additional theoretical properties of wetland community structure--such as inertia, elasticity, amplitude, resilience, hysteresis, malleability, and persistence (to use the terms of Sheehan 1984 and Westman 1978)--that have potential for quantification and testing. However, only a very few experimental studies (e.g., Meffe and Sheldon 1990) have quantitatively examined some of these in a regional set of inland wetlands. If conceptual and operational problems associated with these metrics can be overcome, they may hold potential for more sensitively measuring impacts.

After addressing the question of "How do we express the data?" the next logical question is "What represents normal (or desirable) conditions?" Data interpretation is critical to every monitoring program, and (as discussed in Chapter 2) "normal" can be defined either in terms of (a) the condition of a reference wetland, (b) average regional conditions, or (c) ecological conditions necessary for sustaining the ecosystem type and/or a dynamic balance of its important species. In deference to the vital processes of natural succession that prevail in many wetland types, **a definition of "normal condition" should encompass not only a mean condition, but the naturally-occurring extremes in structure and function that may be expected over decades of time (i.e., temporal and spatial variability).** This report has not sought to go beyond this general consideration and attempt to define nominal (normal) and subnominal (abnormal) wetland conditions. Such an exercise would require an understanding of specific resource management objectives, considerably more data, and significant public involvement.

Finally, if it has been determined that a wetland is "abnormal," it may sometimes be necessary to conclusively determine causality. This typically involves laboratory and field bioassay work, a discussion of which is beyond the scope of this report.

Regardless of which approach is used, caution must be exercised in interpreting community-level data as a potential indication of anthropogenic stress. Absence of a species may be due merely to random events (e.g., Grigal 1985). Sampling metrics, particularly species richness, are often very sensitive to the intensity of sampling, i.e., number of samples, level of effort, size and natural heterogeneity of the wetland sampled. Also, genetic mutation, natural selection, and/or adaptation can result in evolution of tolerant "ecotypes"--local forms of a species that have become tolerant even of normally toxic contaminants. This can alter competitive relationships and ultimately, community structure. Although it is uncertain as to how widespread this phenomenon may be, it can be locally important and has been documented to occur in communities of microbes (Baath 1989), macrophytes (e.g., Christy and Sharitz 1980, McNaughton et al. 1974), aquatic invertebrates (e.g., Krantzberg and Stokes 1989, Kraus and Kraus 1986), and amphibians (e.g., Karns 1984).

Also, the possibility that mobile fish or wildlife are avoiding contaminated areas (even temporarily) should be considered when evaluating community-level vertebrate data. Conversely, wide-ranging biological indicators may not occur even in the "healthiest" wetlands if most other surrounding wetlands have been contaminated or altered.

Finally, **wetland function cannot always be assumed to change whenever the structure of the biological community changes.** Changes in community composition may be compensatory, such that new species replace the function of original species and overall community biomass and perhaps richness does not change (Cairns and Pratt 1986, Herricks and Cairns 1982). An example of this specifically from wetlands is provided by Cattaneo and Kalff (1986), who conducted invertebrate exclusion experiments in an aquatic bed wetland. For this reason, it may be advisable to develop and employ, whenever possible, measurements of both structure and function.

The following sections of this report summarize information relevant to monitoring specific taxonomic groups, wetland types, and stressors. Again, the purpose is not to be prescriptive, but rather to partially survey techniques used by other investigators and summarize conclusions that are relevant to future monitoring. It is expected that these descriptions will be refined and evolve during the review process and as more data are collected from wetlands. The order of these sections does not necessarily reflect priorities, but rather is based on phylogeny (taxonomic relationships). Despite the manner of organization, by major taxa, it is important to recognize that a massive array of interactions can occur in any wetland among the separate taxonomic groups, and such competitive interactions, as noted in a few cases in the following text, can temper the response of an individual taxon to a particular stressor.

4.0 WETLAND MICROBIAL COMMUNITIES

4.1 USE AS INDICATORS

As used here, "microbes" includes bacteria, viruses, yeasts, and microscopic fungi. In wetlands, these have most often been measured indirectly, in the pursuit of estimates of microbe-related processes relevant to element cycling, such as decomposition and denitrification. Although microbial responses to contaminants have been summarized for other surface waters (e.g., Cairns et al. 1972) and upland soils (Baath 1989), few studies have looked at microbial community structure specifically in wetlands, or identified particular microbes as indicators of wetland ecological condition.

Following are discussions of community responses to various stressors. Although we have included some discussion of decomposition rates (an indirect measure of microbial biomass), that process is mainly discussed in Chapter 13.

Enrichment/eutrophication. Microbial abundance and community structure are profoundly affected by trophic status. Enrichment typically results in major increases in microbial abundance (e.g., Tate and Terry 1980) and sometimes richness (Pratt et al. 1989). Enrichment with nitrogen in particular may affect microbial communities, at least in riverine detritus-based systems. Adding nitrogen to streams increased leaf decomposition, microbial biomass, and microbial activity; added phosphorus alone had no effect (Fairchild et al. 1984). Photosynthetic protozoans appear to respond most immediately to nutrient additions (Pratt and Cairns 1985a). However, effects on species richness and community structure have not been extensively studied in most wetland types, and little is known of "indicator taxa" whose use might be most appropriate for signaling enrichment in wetlands.

Microbial colonization rates in a series of shallow Florida ponds was used by Henebry and Cairns (1984) to indicate trophic status. In pond systems (Schmider and Ottow 1985), enrichment increased microbial population densities and number of facultative-anaerobic bacteria (e.g., Streptococci, Enterobacteriaceae and aerobic spore forms, e.g., Bacillus spp., Pseudomonas alcaligenes, and Aeromonas spp.). Mesotrophic ponds had highest numbers of fluorescent pseudomonads. Oligotrophic water had more denitrifiers (Pseudomonas fluorescens and Vibrio spp.).

Organic loading/reduced DO. Given the naturally large organic concentrations in wetlands, it is probable that unique or adapted microbial communities are sometimes present (Felton et al. 1966). Indeed, microbial communities respond strongly to organic additions (Tate and Terry 1980). However, few studies have investigated the effects of increased organic loading and decreased dissolved oxygen on wetland microbial community structure. Low dissolved oxygen (DO) is tolerated or preferred by some taxa, so changes in DO probably trigger significant shifts in community composition.

In cypress domes that received wastewater, Dierberg and Ewel (1984) found faster rates of leaf decomposition (a mainly microbial process). In other surface waters, considerable attention has been focused on coliform bacteria and nuisance growths of Sphaerotilus spp. Large populations of these microbes characteristically develop where sewage has been introduced.

Contaminant Toxicity. The literature concerning response of microbial community structure to heavy metals is summarized by Baath (1989), who includes one study from presumably wetland (organic) soils. That study found a reduction in bacterial abundance at copper concentrations exceeding 275 ug/g. Evidence summarized from non-wetland soils indicates that heavy metal contamination reduces taxonomic richness of the microbial community and causes distinct shifts in taxa; some taxa with potential indicator value are identified by Baath (1989). A shift toward more fungal and gram-negative (vs. gram-positive) taxa may occur, but there is apparently little change in the overall ratio of mycorrhizal to decomposer fungi.

Addition of oils and synthetic organics may result in increased abundance of microbes, particularly species known to degrade and be sustained by petroleum (Walker and Colwell 1977). Microbes, particularly those associated with wetland plants, can be largely responsible for detoxifying some synthetic organic compounds (Hodson 1980) such as pentachlorophenol (Pignatello et al. 1985), and the herbicide glyphosate (brand name, Roundup or Rodeo) (Goldsborough and Beck 1989) as well as detergents (Federle and Schwab 1989).

Thermal Alteration. Although microbial communities are highly sensitive to temperature, few studies have directly examined the effects of thermal stress on community structure in wetlands. In other surface waters, Thermus aquaticus has been found only where heated effluents were introduced (Brock and Yoder 1971).

Acidification. Bogs and other acidic wetlands in many cases contain relatively low richness of microbial taxa (Stout and Heal 1967) and secondary production of microbes can be reduced under such conditions (Benner et al. 1985). However, naturally acidic bogs can have well-adapted, moderately diverse microbial communities (Henebry et al. 1981). Zooflagellate microbes and the ratio of bacteria to fungi can decline with acidification (Leivestad et al. 1976).

Fragmentation of Habitat. We found no explicit information on microbial community response to fragmentation of regional wetland resources. A study of microbial colonization at various distances from an intermittently flooded Virginia wetland (McCormick et al. 1987) found that fewer species colonized introduced substrates that were located a far distance from the wetland; similarity of microbial communities also decreased with increasing distance. One can surmise that as the distance between wetlands with potential microbial colonizers becomes greater, microbial taxa with narrow environmental tolerances and which do not disperse easily might disappear first.

Salinization; Sedimentation/Burial; Turbidity/Shade; Vegetation Removal; Dehydration; Inundation. We found no explicit information on microbial indicators or community response to these stressors in wetlands. From knowledge of microbial responses in other surface waters, it appears likely that microbes in wetlands could respond dramatically to many of these stressors. Undoubtedly data are available from non-wetland surface waters that identify indicator assemblages and document microbial community response to many of these stressors (e.g., Krueger et al. 1988). However, reviewing these was beyond the scope of the present effort, and the transferability of these data to wetlands remains uncertain.

4.2 SAMPLING METHODS AND EQUIPMENT

It is particularly important when using microbes as indicators of anthropogenic disturbance that the comparison wetlands are of about equal age and have similar sedimentary regimes and vegetation densities. This is because microbial communities respond strongly to changes in sediment organic matter, which usually accumulates with wetland age. For example, recently disturbed ponds were found to have fewer microbe species than did natural and older reclaimed ponds on a surface-mined site (Pratt and Cairns 1985b). However, microbial communities in ponds more than two years old were indistinguishable from those in older reclaimed, unreclaimed, and natural ponds despite differences in water quality. Other factors that could be important to standardize among collections of microbial communities include:

> light penetration (water depth, turbidity, shade), temperature, sediment oxygen, baseline chemistry of waters (particularly pH and conductivity), detention time, current velocity, vegetation density, dominant vegetation species, and moisture (e.g., time elapsed since last runoff, inundation, or desiccation event).

Replication requirements for microbial collections are usually significant, due to extremely great spatial and temporal variability of microbial density and diversity. Protozoan "blooms" are more likely to occur in wetlands than in rapidly flowing surface waters, and entirely different communities may exist without apparent cause within millimeters of each other (Carlough 1989). Sampling can occur at any season, but microbial biomass is often greatest in late summer (e.g., Murray and Hodson 1985) and autumn. Standard protocols are available; one is the manual by Britton and Greeson (1988).

Bacterial and fungal abundance are usually estimated as colony forming units (CFU) using plate count techniques. However, concerns have been raised about the validity of this technique for monitoring fungi; use of low-nutrient culture media (rather than the typical enriched media) are also recommended (Baath 1989).

Microbial communities in wetlands are generally collected from sediment samples, water column samples, artificial substrates, or natural organic substrates (e.g., leaf packs). These are described as follows.

Sediment sampling. Sediment sampling of microbial communities can be conducted in all types of wetlands. Dierberg and Brezonik (1982), working in Florida cypress swamps, sampled microbial communities of surface sediments using a sterile piston corer and a plastic syringe with an attached tube.

Water column sampling. Any wetland types that have surface water permanently or seasonally can be sampled using sterile, volumetric containers.

Artificial substrates. Plexiglass plates, acrylic rods, polyurethane foam, or similar inert, sterile surfaces can be placed in any wetlands that have surface water permanently or seasonally, and allowed to be colonized by microbes over a period of several weeks (e.g., Goldsborough and Robinson, 1983, Pratt et al. 1985, Pratt and Cairns 1985b). Substrates are then retrieved and community structure is analyzed. The use of artificial substrates may be a more practical method of sampling protozoa in wetlands than is direct collecting, because of the diversity of microhabitats in wetlands (Henebry and Cairns 1984).

Natural substrates. Natural organic substrates typically contain great numbers of microbes. Consequently, microbial communities have often been collected directly from detrital material, or have been indirectly monitored through measurement of leaf litter decomposition rates. Microbial biomass can also be indirectly monitored by analyzing relative levels of adenosine triphosphate (ATP), e.g., the ratio nM ATP/g ash-free dry weight (Meyer and Johnson 1983). Activity of certain microbial communities was estimated by measuring relative rates of lipid biosynthesis (Fairchild et al. 1984). Adenylate (ATP, ADP, AMP) energy charge ratios in microbes also have been suggested as metrics of ecosystem stress (Witzel 1979).

4.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

In no region of the country, and in no wetland type, have data on microbial community structure been uniformly collected from a series of statistically representative wetlands. Thus, it is currently impossible to state what are "normal" levels for parameters such as seasonal density, species richness, and their temporal and spatial variability. Even qualitatively, lists of "expected" wetland microbial taxa have not been compiled for any region or wetland type.

Limited data suggest that among-wetland variability in microbial community structure is less than variability in vascular plant community structure, but that clear differences exist in microbial communities of marshes, fens, and bogs (Henebry et al. 1981). Microbial density, species richness, and/or colonization rates can be higher in some wetlands than in other surface waters (Duarte et al. 1988, Henebry et al. 1981).

Studies that have compared microbial communities among wetlands (spatial variation) apparently include only Henebry et al. (1981, 1984) and Pratt et al. (1989). The former study, covering 13 Michigan wetlands over a 5-year period, found a range of 93 to 365 protozoan species; Sorenson's similarity index ranged from

0 to 40, with a mean of 21. The latter study, covering 28 Florida ponds, found a range of 112 to 410 species, with a mean of 338 species in non-artificial ponds. Functional group structure of the resident microbial fauna changed slightly from year to year, but wetlands in the same geographic region and experiencing the similar climatic patterns had similar proportions of species in each functional group (Pratt et al. 1989). Microbial densities can vary by 2 to 5 orders of magnitude between sediments, aquatic plants, and the water column (Kusnetsov 1970).

Another study, which examined only one wetland complex (Okefenokee Swamp, Georgia) reported that microbial biomass in sediment ranged from 1 to 28 micrograms gram (dry weight) (Murray and Hodson 1984). A third study, Felton et al.(1967), from Louisiana, reported microbial densities of up to 10^8.

5.0 WETLAND ALGAE

This discussion concerns wetland communities containing phytoplankton, metaphyton, benthic algae, periphyton, and epiphytic algae. Wetlands may contain algal communities that differ from other surface waters, or that indirectly influence community composition of algae in receiving waters. For example, acidic wetland waters commonly are rich in desmid species and acid-tolerant diatoms, such as Eunotia, Frustulia, and Pinnularia (Flensburg and Spalding 1973, Graffius 1958, Patrick 1977). Marshes may become dominated by Nostoc pruniforme, Microcoleus paludosus, Vaucheria sessilis, and sometimes Aphanothece stagnina (Prescott 1968). In a study of the effect on periphyton in a river above and below a marsh, Perdue et al. (1981) found some species of Navicula were common upriver of a marsh but almost non-existent below the marsh; several Nitzschia spp. and Fragilaria spp. were common below but rare above the marsh. Fragilaria construens was abundant in both areas.

5.1 USE AS INDICATORS

As with microbial communities, algal communities in wetlands have most often been measured indirectly, in the pursuit of estimates of photosynthesis, respiration, and productivity. Few studies have quantified algal community structure in wetlands, or identified particular wetland algal species as indicators of wetland ecological condition. However, paleoecological studies of several peatlands have been undertaken. These use diatoms and pollen from peat cores as indicators of ancient environmental conditions (e.g., Agbeti and Dickman 1989, Battarbee and Charles 1987).

Following are discussions of algal community responses to various stressors.

Enrichment/eutrophication and Organic loading . Algal blooms are synonymous with eutrophication, so algae (particularly blue-green forms) are obvious indicators of trophic state, at least in lakes (Hecky and Kilham 1988). As concentrations of phosphorus in flowing water begin to exceed 0.020 mg/L, or 0.015 mg/L (and frequently less) in standing water, significant changes in algal communities can begin to occur (e.g., Traaen 1978), particularly if flow-adjusted loads are greater than 0.22 g/m3 (Craig and Day 1977). Florida regulations for discharge of treated wastewater to forested wetlands specify that, on an annual average basis, waters entering the wetland contain less than 3 mg/L nitrogen and less than 1 mg/L phosphorus; the monthly average for total ammonia must be less than 2.0 mg/L.

Enriched conditions can be associated with either increased (e.g., Morgan 1987) or decreased (e.g., Hooper 1982, Schindler and Turner 1982) species richness of algal communities, depending on whether algae are mostly epiphytic or benthic, the pH, water regime, original state of the system, and other factors. Few studies have used algal community composition to classify the trophic state of wetlands. In other shallow surface waters, taxa such as the following (for example) have become dominant in response to fertilization (Mulligan et al. 1976, Patrick 1977, Prescott 1968):

Anabaena
Aphanizomenon
Closterium
Cosmarium
Dinobryon
Micrasterias
Microcystis
Oscillatoria
Pandorina
Pediastrum
Scenedesmus
Staurastrum
Schroederia

In New Jersey streams exposed to residential and agricultural runoff, Morgan (1987) reported a shift from species characteristic of the region to species that had been geographically peripheral to the region. Algal community structure in some cases might be capable of reflecting the form of enrichment; based on experiments in a Michigan bog, chlorophytean species responded particularly to ammonium, whereas blue-green (cyanobacteria) species dominated when phosphate was added (Hooper 1982). Euglenophytes (one-celled, mobile algae) in particular respond to increases in ammonium and Kjeldahl nitrogen (rather than to nitrate alone), as well as to other substances associated with decomposing organic matter (Hutchinson 1975). Near a wastewater-disposal pipeline in a Michigan bog, several algal species bloomed--Cladophera glomerata, Microspora, Euglena, and Spirogyra (Richardson and Schwegler 1986); algal growth rates were faster at the outfall site than at the control and at various distances away from the outfall.

Contaminant Toxicity. Numerous studies have demonstrated adverse effects of heavy metals (Whitton 1971), herbicides, synthetic organics, oil, and/or heavy metals on freshwater algae. Most such studies have been conducted in laboratories or non-wetland mesocosms, and/or have generally not examined community structure. Several (e.g., Hurlbert et al. 1972) report major algal blooms occurring after insecticide application due to temporary suppression of grazing by aquatic invertebrates. Herbicides have been shown to cause a shift in community composition from large filamentous chlorophyes (green algae) to smaller diatom species and blue-green algal species, particularly those of the order Chaemaesiphonales (Goldsborough and Robinson 1986, Gurney and Robinson 1989, Hamilton et al. 1987, Herman et al. 1986).

Following application of phenol to a shallow pond mesocosm, Giddings et al. (1984, 1985) found and indirectly-caused increase in the dominance of the taxa Euglena, Phacus, Gonium, Coleochaeta, and Scenedesmus. Oil was predicted by Werner et al. (1985) to shift community composition from algae to heterotrophic microbes. In other studies, tolerance to high arsenic levels was demonstrated by Chlorella vulgaris (Maeda et al. 1983) and in a lake contaminated with copper, lead, and zinc, Rhizosoenia eriensis bloomed while other species declined (Deniseger et al. 1990). Algal assays using highway runoff have demonstrated chronic toxicity in several cases, probably due to combined effects of heavy metals, road salt, and sediment (FHWA 1988).

Acidification. Algal responses to acidification in lakes are summarized by Stokes (1981, 1984). Algal species richness can decline in acidified lakes, particularly in the presence of heavy metals (Dillon et al. 1979). Filamentous algae typically show a proportionate increase, and the genus Mougeotia has been reported to be a useful indicator of acidification. Nonetheless, algal production can be relatively high in some naturally acidic wetlands (e.g., Bricker and Gannon 1976).

Thermal Alteration. From knowledge of algal responses in other surface waters (e.g., Squires et al. 1979), it appears likely that algae in wetlands would respond dramatically to thermal effluents, and that suitable assemblages of "most-sensitive species" could eventually be identified.

Dehydration/Inundation. Drawdown of wetland water levels often concentrates nutrients and mobilizes nutrients locked up in exposed peat. This can cause algal blooms in remaining surface water (Schlosser and Karr 1981, Schoenberg and Oliver 1988). Inundation may have the opposite effect, diluting nutrients, reducing nutrient mobilization via oxidation, increasing algal competition with vascular plants, and thus reducing biomass of some algal taxa. However, inundation typically increases the leaf surface area available for colonization by algae, and provides increased opportunities for dispersal of some algal taxa into and out of a wetland. In some Prairie pothole wetlands, metaphyton (unattached, filamentous algae that float in a visible mat) and periphyton (attached algae) increase, while phytoplankton decreases, as higher water levels reduce the density of vascular plants and increase light penetration (Hosseini 1986).

Other Human Disturbance. In other surface waters, species suggestive of "clean" water include Melosira islandica and Cyclotella ocellata. Algal or microbial species that can indicate "contaminated" water include Chlamydomonas, Euglena viridis, Nitzschia palea, Microcystis aeruginos, Oscillatoria tenuis, O. limosa, Stigeocloneum tenue, and Aphanizomenon flos-aquae (Prescott 1968, APHA 1980).

Salinization; Sedimentation/Burial; Vegetation Removal; Fragmentation of Habitat. We found no explicit information on algal indicators or algal community response to these stressors in wetlands. From knowledge of algal responses in other surface waters (e.g., Dickman and Gochnauer 1978), it appears likely that algae in wetlands would respond dramatically to many of these stressors, and that suitable assemblages of "most-sensitive species" could be identified.

5.2 SAMPLING EQUIPMENT AND METHODS

Factors that could be important to standardize (if possible) among collections of algal communities include:

> age of wetland (successional status), light penetration (water depth, turbidity, shade), hydraulic residence time, temperature, conductivity and baseline chemistry of waters, current velocity, leaf surface area and stand density of associated vascular plants, density of grazing aquatic invertebrates, typical duration and frequency of wetland inundataion, and time elapsed since last runoff or inundation event.

Standard protocols for algal monitoring are available (Aloi 1990, Britton and Greeson 1988), although uncertainty exists concerning their applicability to wetlands.

Replication requirements in wetland algal studies are significant, due to large spatial and temporal variability. Some investigators have recommended that samples that will be assumed to come from the same time period should be sampled within a time period less than the hydraulic residence time of the wetland. Rapid succession in dominant flagellate species was typical of shallow, eutrophic ponds where conditions fluctuate quickly (Estep and Remsen 1985).

Sampling can occur at any season, but algal biomass is often greatest during the mid to late growing season (e.g., Crumpton 1989, Hooper 1978, Hooper-Reid and Robinson 1978a, b). In deeper waters, it may be advisable to sample phytoplankton at mid-day, due to vertical movements at other times (Estep and Remsen 1985). The pigment, chlorophyll-a is sometimes sampled from the water column as an indicator of algal biomass, but yields little information on community structure. Rabe and Gibson (1984) found greater phytoplankton density in a shallow vegetated pond than at nonvegetated sites, but species composition was similar. In contrast, Seelbach and McDiffett (1983) found that a pond with submerged vegetation had more taxa but lower population density than an open-water pond.

Algal communities in wetlands are generally collected from sediment samples, water column samples, artificial substrates, or natural organic substrates. Methods are described as follows.

Sediment sampling. Algae can be sampled from sediment surfaces in all types of wetlands. Piston corers, plastic syringes, or other suction devices are typically used.

Water column sampling. Any wetland types that have surface water permanently or seasonally can be sampled. Samples from surface waters commonly involve use of volumetric containers or fine-mesh nets. Vertically-integrating, automated samplers can be used (e.g., Schoenberg and Oliver 1988). Surface microlayers (top 250-440 micrometers) can be sampled using fine nets or screens mounted on a frame (e.g., Estep and Remsen 1985). In flowing-water wetlands, fine nets can be mounted to intercept algae carried by currents.

Artificial substrates. Artificial substrates (initially sterile materials placed in a wetland and subjected to natural colonization) may integrate algal assemblages from a large variety of microhabitats. As with microbial communities, algal communities can be monitored by installing plexiglass plates or similar inert, sterile surfaces in any wetlands that have surface water permanently or seasonally, and allowing them to be colonized by attached algae over a period of several weeks. Substrates are then retrieved and community structure is analyzed (e.g., Hooper-Reid 1978).

Natural substrates. Natural organic substrates, particularly those in shallow water, may contain a great biomass of algae. Epiphytic and epibenthic algae are often sampled using a quadrat approach, in which a frame is placed over a standard-sized area of bottom or a standard volume of the water column is enclosed. Frame sizes of 10 x 10 cm (Atchue et al. 1983) and 1-2 m^2 (Schoenberg and Oliver 1988) have been used.

If algal density is to be estimated accurately, the surface area of substrate must be quantified. This can be a daunting task in the case of epiphytic algae, where plant surface areas need to be measured. Some investigators have approached this by measuring surface areas of a random sample of plants, sometimes with the use of a digital scanner, then measuring their volumes (by displacement) or dry weights and developing area-volume or area-weight calibration curves. The curves can be used to estimate plant surface area from future, simpler measurements of the volume or weight of other plants of the same species.

5.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

In no region of the country, and in no wetland type, have data on algal community structure been uniformly collected from a series of statistically representative wetlands. Thus, it is currently impossible to state what are "normal" levels for parameters such as seasonal density, species richness, and their temporal and spatial variability.

Studies that have compared algal community structure among wetlands (spatial variation) apparently include only Hern et al. (1978) who studied the Atchafalaya system in Louisiana, and Sykora(1984), who reported a range of 9 to 21 phytoplankton taxa per ml (mean=9, S.D.=2.3) from a series of six West Virginia wetlands. Phytoplankton density (cells per ml) ranged from 19 to 2581 (mean=203, S.D.=126). Atchue et al. (1982) found 56 taxa of phytoplankton in 8 springtime collections from a one-hectare temporary swamp pool in Virginia. We encountered no journal papers that quantified measurement errors or year-to-year variation in microbial community structure in U.S. inland wetlands.

Even qualitatively, lists of "expected" wetland algal taxa appear not to have been compiled for any region or wetland type. Limited qualitative information may be available by wetland type from the "community profile" publication series of the U.S. Fish and Wildlife Service (USFWS)(Appendix C).

6.0 NON-WOODY (HERBACEOUS) VEGETATION

This discussion concerns communities dominated by mosses, lichens, liverworts, ferns, sedges, etc., and includes emergent, floating-leaved, and submersed forms. These taxa probably have been studied more than any other wetland taxa.

6.1 USE AS INDICATORS

Following are discussions of the community-level responses of herbaceous vegetation to various stressors.

Enrichment/eutrophication. Species richness of herbaceous plants, particularly emergent species, can increase with moderate enrichment (Graneli and Solander 1988). However, severe enrichment drastically shifts community structure, and can decrease species richness (e.g., Lachavanne 1985, Lind and Cottam 1969, Tilman 1987, Hough et al. 1989, Toivonen and Back 1989). This might be particularly true of macrophyte communities in flowing water wetlands (e.g., Pip 1987), where nutrients otherwise tend to be less limiting than in most standing water (basin) wetlands. Duarte and Kalff (1988), studying lacustrine macrophytes, similarly found that the effect of fertilization was influenced by hydrologic energy (e.g., wave action).

The greatest richness of emergent plants has been reported to occur when standing biomass of the community is less than 1000 g/m^2 in British wetlands, 400-500 g/m^2 in Netherlands wetlands (Vermeer and Berendse 1983), and 60-500 g/m^2 in Ontario wetlands. If a goal is to maintain within-wetland species richness, the particular nutrient loadings that result in a desired biomass might be calculated from empirical data (e.g., Duarte and Kalff 1986, Duarte et al. 1986) to derive very approximate criteria for nutrient loadings, and perhaps, with further testing, for other factors that can increase plant biomass (e.g., thermal warming, hydrologic regime). However, the numeric ranges just given are probably less valid for wetlands that are grazed or subject to other significant vegetation removal processes.

Changes in composition and growth of herbaceous communities as a probable result of increased nutrients have been reported by many ecologists, including:

> Guntenspergen 1984, Haslam 1982, Kullberg 1974, Jensen 1979, Kadlec and Hammer 1980, Klowsowski 1985, Kohler et al, Mahoney 1977, Pringle and van Ryswyck 1968, Schwartz and Gruendling 1985, Seddon 1972.

An important regional impact of excessive enrichment is that small, regionally rare plant species (that often characterize infertile wetlands or wetlands whose chemistry reflects weak buffering) are often out-competed by large, regionally common species (Day et al. 1988, Moore et al. 1989). Insectivorous plants, quillworts, many species that typify fen wetlands, and some orchids and mosses that typify oligotrophic wetlands, are particularly sensitive to enrichment, either airborne or waterborne (Moore et al. 1989, Roelofs 1983, 1986, Schuurkes et al. 1986). Percent cover of the dominant peat-forming mosses of bogs can probably be reduced by atmospheric nitrogen deposition rates of 4.3 $g/m^2/yr$, but not 2.0 $g/m^2/yr$ (Ferguson et al. 1984). However, because great variability exists among tolerances of moss species, a limit of 2.0 and possibly (in oligotrophic wetlands) 1.0 $g/m^2/yr$ has been suggested (Schuurkes et al. 1987, Liljelund and Torstensson 1988).

Submersed and floating-leaved or mat-forming species usually respond more strongly to enrichment than do emergents (e.g., Ozimek 1978, Shimoda 1984), because the former obtain nutrients directly from the water column, whereas the latter obtain them from sediments. In many regions, vascular floating-leaved plants such as pondweed (Nuphar), duckweed (Lemna), and water-meal (Wolffia) become more prevalent with increasing enrichment (e.g., Bevis and Kadlec 1990, Burk et al. 1976, Ewel 1979, Kadlec et al. 1980), and in severely eutrophic lakes, emergent species may survive as floating mats (Graneli and Solander 1989).

Species shifts may be less immediate or noticeable when moderate amounts of nutrients are added to wetlands that are already eutrophic, because high microbial populations that characterize such environments can be highly effective at first in competing for the nutrients (e.g., Richardson and Marshall 1986). With extreme enrichment, submersed macrophytes can eventually decline, probably as a result of being shaded out by algae (e.g., Mulligan et al. 1976, Phillips et al. 1978), and emergent species may increase.

Among emergent species, extreme eutrophication causes decreased species richness because (a) turbidity of phytoplankton blooms shades out many submersed species, and (b) as phytoplankton decays, resultant oxygen deficits in bottom sediments probably stress the most sensitive rooted species (e.g., Hartog et al. 1989). Of the emergent species, cat-tail (Typha) and common reed (Phragmites) often dominate enriched wetlands and may be the least sensitive to the initial stages of eutrophication (e.g., Kadlec 1979, Hartland-Rowe and Wright 1975, Kadlec 1990, Kadlec and Bevis 1990, Moore et al. 1989). Cat-tail biomass and production respond to annual fluctuations in nitrate, making cat-tail a successful opportunist capable of dominating wetlands that have erratic inputs of nutrients (Davis 1989). Although Phragmites can exist without any obvious sign of harm in wetlands with at least 6 mg/L phosphorus and 10 mg/L nitrogen (Ostendorp 1989), massive die-offs of this species in European wetlands have been attributed by some to excessive enrichment (Hartog et al. 1989, Ostendorp 1989). Another emergent plant--manna grass (Glyceria grandis)--increased in dominance in an Ontario wetland subjected to treated effluent (Mudroch and Capobianco 1979).

In Michigan, moderately eutrophic lakes were dominated by Ceratophyllum demersum, Utricularia vulgaris, and Cladophora fracta (Hough et al. 1989). In England, Potamogeton pectinatus, Myriophyllum spicatum, and Hippuris vulgaris dominate in highly eutrophic waters (Butcher 1946, Seddon 1971). However, Kadlec et al. (1980) and Mulligan et al. (1974) found Myriophyllum declined under increasing fertilization, along with Ceratophyllum demersum, Polygonum and Utricularia. Many wetland plant species are categorized by their nutrient-level preferences, and thus as their potential as indicators of eutrophication, in reports by Ellenberg Jeglum 1971, Moyle 1945, Pip 1979, Stewart and Kantrud 1972, Swindal and Curtis 1957, and Zoltai and Johnson 1988.

Even the submersed types of herbaceous vegetation appear a poorer indicator of eutrophication than are algal communities, which respond more quickly (Crumpton 1989). Neither macrophyte nor algal taxa are reliable indicators of moderate enrichment in naturally enriched waters, e.g., minerotrophic fens, wetlands in karst limestone regions (Hellawell 1984, Strange 1976).

Organic loading/reduced DO. Existing literature often does not adequately distinguish the effects on herbaceous plants of organic loading/reduced DO, from the effects of nutrients (discussed above) or inundation (discussed below).

At least in the short term, the biomass of herbaceous plants generally increases with moderate additions of wastewater. In acidic, oligotrophic wetlands (e.g., bogs), species richness may increase (e.g., Guntenspergen 1984). Community components with short turnover times, such as aboveground biomass and leaf area of annual plants, can respond most sensitively (e.g., Brown 1981, Odum et al. 1984).

Aggressive, introduced annuals sometimes replace native perennial species (e.g., Finlayson et al. 1986). While the occurrence of rarer, perennial species is often correlated with specific chemical conditions, the occurrence of aggressive, common species often is not (Pip 1979). Populations of such species tend to be more plastic in their response to wastewater enrichment (e.g., Guntenspergen 1984).

Over longer periods of time and/or excessive loading, wastewater additions may result in stress from low dissolved oxygen, increased hydrogen sulfide, and excessive accumulation of sediment organic matter. These conditions can selectively inhibit certain plant taxa (Barko and Smart 1983), particularly those that are unable to translocate oxygen to their roots (Brennan 1985). While cattail (Typha) require only trace amounts of dissolved oxygen for germination (Leck and Graveline 1979), bud development is more successful in reeds (Phragmites) if flooded soils are aerated (Haslam 1973), as is sprouting of purple nutsedge (Cyperus rotundus) (Al-Ali et al. 1978).

Morgan and Philipp (1986) surveyed a host of New Jersey streams and listed 22 species found only in streams that, based on their location and limited chemical sampling, were assumed to be "polluted." The researchers found 18 only in "unpolluted" streams, and 21 in both types. Callitirche heterophylla, Ludwigia palustris, Polygonum punctatum, Potamogeton epihydrus, and Sparganium americanum were locally dominant only in polluted streams, and Sagittaria englemanniana, Scirpus subterminalis, and Vaccinium macrocarpon were dominant only in unpolluted streams. Polluted sites, with high nitrate and pH, had a higher percentage of non-indigenous species, vines, and herbaceous (vs. woody) plants. Vines and other low-growing species also were found by Nilsson and Grelsson (1990) to dominate riverine sites with intermediate accumulations of organic matter (i.e., 100-200 g/m^2 leaf litter), whereas sites with very low or very high accumulations of organic matter were dominated by stemmed species. Emergent plant species richness also showed such a quadratic correlation with accumulated organic matter.

Contaminant Toxicity. Some herbaceous plants are quite sensitive to heavy metals and other contaminants, and as a result, contamination can alter species composition, and decrease species richness, canopy coverage, and net annual productivity of wetland communities (e.g., Cooper and Emerick 1989, Olson 1979). Based on studies of eight Colorado wetlands exposed to varying degrees of heavy metal-contaminated runoff, Cooper and Emerick (1989) noted:

> "Subalpine fen wetlands in the Colorado Front Range that have less than 3 vascular plant species growing in the main part of the wetland (not the edges) and have less than 50 percent total canopy coverage and less than 100 g/m^2 total annual primary production, are likely to indicate impact from heavy metal toxicity. An exception is areas that are flooded or have ponded water for much of the growing season."

Forbs (herbaceous dicots in that study) seemed particularly uncommon in polluted wetlands. The authors noted no species that occurred only at contaminated sites, but found that the sedges, Carex aquatilis, C. utricularia, and/or C. scopulorum, predominated in these areas. Species absent from areas contaminated by large concentrations of heavy metals included the following:

Swertia perennis
Caltha leptosepala
Geum macrophyllum
Sedum (Clementsia) rhodantha
Bistorta bistortoides
Polygonum (Bistorta) bistortoides and vivipara
Cardamine cordifolia
Epilobium lactiflorum
Galium trifidum
Juncus albescens
B. vivipara

Duckweed (Lemna) is particularly sensitive to the heavy metals cadmium and nickel, and chromium concentrations of 10 mg/L are inhibitory (Huffman and Allaway 1973). Cattail (Typha latifolia) can tolerate lead, copper, and chromium accumulations of at least 10 micrograms/g dry weight of aboveground biomass; zinc accumulations in cattail may reach 25 micrograms/g dry weight without apparent ill effects (Mudroch and Capobianco 1979). The common reed (Phragmites) can tolerate industrial wastewater with high levels of heavy metals (e.g., up to 250 micrograms/g sediment copper concentrations), as do bulrushes (e.g., Seidel 1966).

Heavy metals and other toxicants borne in air currents and precipitation have widely been reported to alter community composition of mosses and lichens (e.g., Lee et al. 1987, Sigal and Nash 1980). Species of mosses and lichens differ considerably in their sensitivity to metals, and are prevalent in many wetland types. Thus, they may have considerable potential for use as indicators of this type of pollution.

A decline in Asclepias syriaca (milkweed) and an overall increase in species richness and equitability may have been related to contaminants associated with incinerator residue deposited in an emergent marsh in Massachusetts (Mika et al. 1985). In a major Ohio river, Stuckey and Wentz (1969) reported the following species to be rare or absent from waters contaminated by industrial effluents, but common in analogous uncontaminated habitats:

Justicia americana
Saururus cernuus
Phytostegia virginiana
Rumex verticillatus
Samolus parviflorus
Carex frankii
Lycopus rubellus
Lippia lanceolata
Helenium autumnale
Eclipta alba
Scirpus americanus
Amaranthus tuberculatus
Hibiscus militaris
Strophostyles helvola

Also, these investigators found the following plants to be common in industrially contaminated waters:

Polygonum hydropiper
P. persicaria
P. pensylvanicum
P. coccineum
P. lapathifolium
P. punctatum
Sagittaria latifolia
Echinochloa pungens
Leersia oryzoides
Ambrosia trifida
Urtica dioica
Arctium minus
Bidens frondosa

In an Ontario river, submersed species (Elodea, Ceratophyllum, and Myriophyllum) appeared to be less tolerant of industrial wastes than floating-leaved and short, rooted aquatic plants (Potamogeton, Nuphar, and Nymphaea), which were in turn less tolerant than cattail (Typha) and common reed (Phragmites)(Dickman et al. 1980, 1983, Dickman 1988).

Floating-leaved herbaceous plants are sensitive to the physical effects of oil, and growth of the duckweed Spirodela oligorhiza is affected by PCB concentrations of 5 mg/L (Mahanty 1975). Cattail can tolerate petroleum oil concentrations of 1 g/L (Merezhko 1973) and, along with common reed (Phragmites), appeared to be the most tolerant macrophyte downstream from an industrial effluent source in Ontario (Dickman 1988). The response of wetland species to an oil spill in a Massachusetts inland wetland (Burk 1977) was as follows (* = annual species):

Species not recorded after oil spill:

Bidens cernua*
B. connata*
B. frondosa*
Echinochloa walteri*
Eleocharis obtusa
Galium tinctorium*
Hypericum mutilum
Spirodela ployrhiza
Verbena hastata
H. virginicum
Iris versicolor
Lycopus uniflorus
Mimulus ringens
Polygonum punctatum*
P. sagittatum*
Sparganium americanum
Vallisneria americana

Species reduced after oil spill:

Cephalanthus occidentalis
Eleocharis acicularis
Najas flexilis*
Onoclea sinsibilis

Galium trifidum
Leersia oryzoides
Lindernia dubia*
Ludwigia palustris
Pilea fontana*
Pontederia cordata
Scirpus pedicellatus
Zizania aquatica

Species apparently unaffected or increasing after oil spill:

Alisma subcordatum
Carex lurida
Ceratophyllum demersum
Dulichium arundinaceum
Eleocharis palustris
Elodea nuttallii
Equisetum fluviatile
Lemna minor
Lysimachia terrestris
Nuphar variegatum
Veronica scutellataq
Polygonum coccineum
Potamogeton crispus
P. epihydrus
Sagittaria graminea
S. latifolia
Salix nigra
Scirpus cyperinus
S. validus
Scutellaria leteriflora
Sium suave
Vitis labrusca

Bulrushes are killed by phenol concentrations of 100 mg/L and abnormalities occur at large phenol concentrations, but new shoots form quickly (Seidel 1966). Herbicides have often been used to control some herbaceous species, notably purple loosestrife (Lythrum) and common reed (Phragmites), and undoubtedly affect some non-target species as well. However, herbicide effects can be species-specific, with the result being that some applications result in overall increase in algae and plant richness (although perhaps lower overall productivity), as monotypic or dominant stands are opened for invasion by less aggressive species (e.g., Murphy et al. 1981). Detergent concentrations of 15 mg/L can damage wetland macrophytes (Agami et al. 1976).

Additional toxicological information may be available through EPA's PHYTOTOX (Royce et al. 1984) and AQUIRE databases.

Acidification. Ambient pH is one of the most important factors affecting community composition of emergent and aquatic bed wetlands bordering northern lakes (Hultberg and Grahn 1975), as well as peatlands (e.g., Anderson 1986, Jeglum 1971) and perhaps other low-alkalinity, standing water wetlands. It can be a stronger influence in these systems than nutrient status or water transparency (e.g., Jackson and Charles 1988). However, its effect on overall species richness is unclear (Eilers et al. 1984, Jackson and Charles 1988, Yan et al. 1985,). Usually, fewer species of macrophytes are found in acidic lakes than in circumneutral lakes (e.g., Friday 1987, Hunter et al. 1986, Hutchinson et al. 1985), but these are often species that are regionally rare (Moore et al. 1989).

The study of Adirondack (New York) lacustrine wetlands by Jackson and Charles (1988) reported the following taxa to be relatively intolerant of acidification: Najas flexilis, Nitella flexilis, Potamogeton pusillus, P. natans, and P. amplifolius. Submersed and floating-leaved species present at pH lower than 5.5 but not in less acidic conditions included Potamogeton confervoides and Sparganium angustifolium; species present in both acidic (pH < 5.0) and circumneutral wetlands included Nuphar, Juncus pelocarpus, Drepanocladus fluitans, Utricularia vulgaris, Isoetes muricata, Eriocaulon septangulare, Sagittaria graminea, and Myriophyllum tenellum (Jackson and Charles 1988). Emergent species present in both acidic (pH < 5.0) and circumneutral wetlands included Calla pallustris, Juncus brevicaudatus, Dulichium arundinaceum, Lysimachia terrestris, and Juncus pelocarpus (Jackson and Charles 1988). Wolffia, Lemna, and Spirodela have optimal pH's of 5.0, 6.2, and 7.0 respectively, whereas their tolerated ranges are (respectively) 4 - 10, 4 - 10, and 3 -10 (McClay 1976).

In some northern wetlands, especially those that are heavily shaded, acidification can result in increased presence of mat-forming mosses of the genus Sphagnum (e.g., Gignac 1987, Grahn 1976, Roberts et al. 1985), and these mosses can further lower the pH (e.g., Glime et al. 1982). However, under severe acidification and accompanying deposition of industrial pollutants, Sphagnum can decline and in some cases, be replaced by cottongrass (Eriophorum)(e.g., Gorham et al. 1987, Lee et al. 1987a).

Cattail, rushes, and sedges occur in sediments with a pH of at least 4.7 (Dykyjova and Ulehlova 1978), while common reed and nutsedge can tolerate a pH as low as 2.0 (Al-Ali et al. 1978, Dykyjova and Ulehlova 1978). Natural stands of sedge (Carex) have a pH range from 4.9 to 7.4 (Baker 1971), while the range for reed canary-grass (Phalaris) is 6.1 to 7.7 (Gross and Jung 1978, Dean and Clark 1972, Niehaus 1971, Allinson 1972). Many regional botanical texts describe approximate pH ranges of individual wetland species (e.g., Crow and Hellquist 1981), as does some literature not excerpted here (e.g., Jeglum 1971, Swanson 1988).

Reductions in plant species diversity, decreased productivity, and life cycle disruptions were among the effects attributed to high pH values downslope from a Massachusetts hazardous waste lagoon (USEPA 1989a).

Salinization. Saline inland wetlands commonly have fewer species of macrophytes (Pip 1979, Reynolds and Reynolds 1975), and may be particularly deficient in species that typically form floating mats (Lieffers 1984). Most freshwater macrophytes cannot tolerate more than 10 ppt dissolved salts (Reimold and Queen 1974). Inland wetland plants that reportedly tolerate specific conductivity of greater than about 5 mS/cm are shown in Table 7, from Kantrud et al. (1989). Other data on salinity tolerances of inland wetland plants are provided by Reimold and Queen (1974) and others listed in Table 7.

Contamination of a northern Indiana bog with road salt resulted in almost complete elimination of endemic species and replacement by non-bog species, dominated by Typha angustifolia (Wilcox 1987), which can sometimes tolerate salinities of up to 25.5 ppt (Philipp and Brown 1965, Shekov 1974), at least for short periods. As salt concentrations declined in the four years of the study, endemic plants began to recolonize the affected area; biomass and growth of Sphagnum fimbriatum was significantly reduced at NaCl concentrations greater than 900 mg/L Cl- (Wilcox 1987). The common reed (Phragmites communis) tolerates salinities of up to 45 ppt, although seedlings may be killed by salinities of 10 ppt. Duckweed (Lemna minor) has reduced growth at salinities above 7 ppt (Haller et al. 1974, Stanley and Madewell 1976). For many species, these values vary by genetic population, life stage, duration of exposure, temperature, and other factors. The freshwater cattail, Typha latifolia, as expected, is less salinity-tolerant than the estuarine cattail, Typha angustifolia, mentioned above (McNaughton 1966). However, a presumed hybrid, Typha gauca, appeared resistent to road salt runoff (Bayly and O'Neil 1972). Even Typha latifolia seeds appeared more tolerant of road salt in snowmelt than germinating wool-grass (Scirpus cyperinus) and three-way sedge (Dulichium arundinaceum); purple loosestrife seeds (Lythrum salicaria) were similarly tolerant (Isabelle et al. 1987). The rush, Scirpus acutus, appears more salt-tolerant than its many of its congeners (Smith 1983).

Table 7. Examples of Aquatic Macrophytes Tolerant of Saline Conditions in Inland Wetlands.

These lists are reproduced from Kantrud et al. (1989), and deal primarily with prairie pothole wetlands; applicability to other wetland types is unknown. Additional salt-tolerant species may be found in lists of Haller et al. 1974, Kauskik 1963, Lieffers 1984, Mall 1969, McKee and Mendelssohn 1989, Millar 1976, Millar 1978, Moyle 1945, Nelson 1954, Pip 1979, 1987a,b, Reynolds and Reynolds 1975, and Stewart and Kantrud 1972.

	Specific conductivity (mS/cm)[a]		
Species	Mean	Min.	Max.
Vernonia fasciculata	0.1	0.1	0.2
Agrostis stolonifera	0.2	--	--
Lycopus americanus	0.3	--	--
Potentilla rivalis	0.3	--	--
Carex stipata	0.4	--	--
Equisetum arvense	0.4	--	--
Juncus interior	0.4	0.3	0.9
Aster sagittifolius	1.0	--	--
Plantago major	1.0	--	--
Potentilla norvegica	0.3	0.1	1.1
Juncus dudleyi	0.4	0.3	1.3
Carex buxbaumii	1.2	1.0	1.4
Lysimachia hybrida	0.1	0.1	1.6
Carex vulpinoidea	1.0	0.1	1.7
Ranunculus macounii	1.1	0.1	2.1
Rumex mexicanus	0.5	0.1	2.2
Juncus bufonius	2.3	--	--
Cirsium arvense	2.5	--	--
Bidens cernua	1.5	0.7	2.5
Helenium autumnale	1.5	0.5	2.5
Carex praegracilis	0.3	0.1	3.0
Echinochloa crusgalli	1.3	0.5	3.2
Carex laeviconica	1.5	0.1	3.2
Rorippa islandica	1.7	0.1[b]	3.2
Poa palustris	1.4	Tr.	3.4
Stachys palustris	1.8	0.1	3.6
Calamagrostis canadensis	1.4	0.4	3.8
Carex sartwellii	1.5	0.4	3.8
Lycopus asper	1.9	0.4	4.4
Epilobium glandulosum	1.5	0.5	4.7
Mentha arvensis	1.6	0.1	4.9
Apocynum sibiricum	1.8	0.4	5.0
Eleocharis compressa	2.0	0.7	5.0
Carex tetanica	2.0	0.9	5.5
Potentilla anserina	1.6	0.1	6.0
Boltonia asteroides	1.4	0.1	6.8
Carex lanuginosa	2.0	0.1	9.1
Teucrium occidentale	3.1	0.2	9.1
Aster hesperius	2.4	0.4	9.8
Juncus torreyi	1.7	0.2	10.0
Aster simplex	1.8	0.1	16.1
Calamagrostis inexpansa	2.6	Tr.	17.6
Juncus balticus	3.3	0.1	20.1

Table 7 continued

Species	Specific conductivity (mS/cm)[a] Mean	Min.	Max.
Spartina gracilis	9.0	0.7	20.1
Plantago eriopoda	9.8	1.0	20.1
Sonchus arvensis	5.2	0.5	20.8
Spartina pectinata	3.0	Tr.	33.5
Muhlenbergia asperifolia	11.0	0.7	38.5
Hordeum jubatum	7.8	Tr.	48.6
Triglochin maritima	12.5	0.7	50.9
Distichlis spicata	17.0	0.5	76.4
Atriplex patula	23.0	6.9	76.4

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[a] Underlined means (Disrud 1968; Kantrud et al. 1989) indicate surface water measurements in wetlands where the species reached peak abundance; underlined ranges (ibid) are for instances where the species occurred in waters of greater or lesser salinity than that recorded by Smeins (1967).

[b] Indicates measurements <0.05 mS/cm.

Table 7 continued

Species	Water regime	Specific conductivity (mS/cm)[a]		
		Mean	Min.	Max.
Equisetum fluviatile	SE	0.3	--	--
Galium trifidum	SA	0.3	--	--
Scutellaria galericulata	SA	0.3	--	--
Impatiens biflora	SA	0.4	--	--
Mimulus ringens	SA	0.6	--	--
Eupatorium maculatum	SA	0.7	--	--
Sagittaria cuneata	SE	0.7	--	--
Glyceria striata	SA	0.8	--	--
Ranunculus gmelini	SA	0.8	--	--
Asclepias incarnata	SA	0.8	0.8	0.9
Parnassia glauca	SA	0.9	--	--
Glyceria borealis	SE	1.0	--	--
Salix interior	SA	0.4	0.3	1.7
Carex lacustris	SA	1.4	0.9	1.7
Solidago graminifolia	SA	0.8	0.1	2.1
Polygonum amphibium	SE	0.6	0.1	2.2
Scirpus atrovirens	SA	1.0	0.5	2.2
Cicuta maculata	SA	1.4	0.5	2.2
Eriphorum angustifolium	SA	1.8	0.5	2.2
Carex rostrata	SA	1.1	0.2	2.6
Polygonum coccineum	SE	1.3	Tr.[b]	3.4
Phalaris arundinacea	SE	1.6	0.1	3.8
Carex aquatilis	SA	1.6	0.3	3.8
Lysimachia thrysiflora	SA	1.7	0.5	3.8
Glyceria grandis	SE	0.7	Tr.	4.0
Sium suave	SE	1.8	0.1	4.0
Scirpus heterochaetus	SP	1.4	0.1	4.2
Alopecurus aequalis	SE	1.1	0.1	4.5
Sparganium eurycarpum	SE	1.8	Tr.	4.6
Eleocharis acicularis	SE	1.5	0.1	5.8
Scirpus validus	SP,SA	1.8	0.2	6.2
Typha X *glauca*	SP	0.8	0.1	6.6
Sagittaria cuneata	SE	1.8	0.1	6.7
Scirpus fluviatilis	SP	1.9	0.3	6.7
Alisma gramineum	SE	2.0	0.3	6.7
Carex atherodes	SE	2.0	Tr.	8.5
Ranunculus sceleratus	SE	3.6	0.1	8.5
Beckmannia syzigachne	SE	1.5	Tr.	9.5
Alisma plantago-aquatica	SE	1.8	Tr.	9.5
Ranunculus cymbalaria	SE	3.5	0.6	9.5
Scolochloa festucacea	SE	3.4	0.1	12.1
Phragmites australis	SA	3.5	0.5	12.4
Typha latifolia	SP,SA	2.1	0.1	13.6
Typha angustifolia	SP	3.4	0.4	13.6
Eleocharis palustris	SE	2.7	0.1	14.5
Scirpus nevadensis	SE	15.7	12.0	20.5
Scirpus acutus	SP	4.3	0.2	24.0
Suaeda depressa	SE	24.0	5.0	66.0
Scirpus americanus	SE	4.9	0.5	70.0
Scirpus maritimus	SP	10.3	0.8	76.4
Puccinellia nuttalliana	SE	20.0	1.4	76.4

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[a] Underlined means (Disrud 1968; Kantrud et al. 1989) indicate surface water measurements in wetlands where the species reached peak abundance; underlined ranges (ibid) are for instances where the species occurred in waters of greater or lesser salinity than that recorded by Smeins (1967).

[b] Indicates measurements <0.05 mS/cm.

SA=semiannual, SE=seasonal, SP=semipermanent

Table 7 continued

Species	Specific conductivity (mS/cm)[a] Mean or single measurement	Min.	Max.
Myriophyllum pinnatum	0.1	--	--
Nuphar variegatum	0.4	--	--
Najas flexilis	0.4	--	--
Elodea canadensis	0.4	--	--
Potamogeton friesii	0.6	0.3	1.0
Myriophyllum verticillatum	1.1	--	--
Potamogeton gramineus	0.8	0.1	1.2
Callitriche palustris	0.3	0.1	1.7
Hippuris vulgaris	0.7	0.5	2.3
Callitriche hermaphroditica	1.2	0.5	2.5
Ranunculus flabellaris	2.0	0.4	2.5
Potamogeton zosteriformis	1.0	0.3	2.8
Spirodela polyrhiza	1.5	0.6	3.0
Ricciocarpus natans	2.2	0.3	3.2
Drepanocladus spp.	1.2	0.1	3.3
Potamogeton vaginatus	3.3	--	--
Potamogeton richardsonii	1.7	0.3	4.0
Ranunculus subrigidus	1.4	0.2	4.5
Riccia fluitans	2.1	0.1	4.7
Ceratophyllum demersum	2.1	0.1	5.1
Potamogeton pusillus	2.1	0.1	6.7
Myriophyllum spicatum	2.2	0.2	6.7
Utricularia vulgaris	2.7	0.1	8.1
Lemna minor	3.1	0.1	10.9
Lemna trisulca	3.2	0.1	13.9
Ruppia maritima var. *occidentalis*	4.1	0.6	14.2
Zannichellia palustris	4.8	0.3	25.0
Chara spp.	2.2	0.3	42.0
Potamogeton pectinatus	6.5	0.4	60.0
Ruppia maritima var. *rostrata*	36.1	5.5	66.0

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[a] Underlined means (Disrud 1968; Kantrud et al. 1989) indicate surface water measurements in wetlands where the species reached peak abundance; underlined ranges (ibid) are for instances where the species occurred in waters of greater or lesser salinity than that recorded by Smeins (1967).

Sedimentation/Burial. We found little explicit information on overall macrophyte community response to burial or sedimentation. In some cases, sedimentation creates shoals in rivers or lakes, which provide sufficient substrate within the euphotic zone for herbaceous wetlands to become established or expand, at least until a major scouring flood re-occurs (e.g., Burton and King 1983). Where sedimentation is severe, water may become too shallow for some submersed species and a shift to emergent species may occur (Edwards 1969).

Differences probably exist among herbaceous plant species with regard to their intrinsic tolerance and adaptability to excessive sedimentation. Species often noted as occurring in disturbed, sediment-laden wetlands include the common reed (Phragmites), reed canary-grass (Phalaris)(Reed et al. 1977), and other large and robust taxa. Repeated burial by as little as 5 cm of sediment per year can be detrimental to some emergent species (van der Valk et al. 1981).

Shading/turbidity. Increased shade or turbidity (whether from suspended sediment, phytoplankton, natural staining, or other sources) generally results in a shift in community structure from submersed species to floating-leaved or emergent species (Hough and Forwall 1988). Turbidity increases and decreases in bank stability may also favor an increase in the proportion of invasive, dominating species to the exclusion of less aggressive native macrophytes (Morin et al. 1989).

A 25 NTU (nephelometric turbidity units, or about 100 mg/L suspended solids) increase in turbidity in a shallow riverine wetland can reduce production of algae and submerged aquatics by 50 percent (Lloyd et al. 1987) and a mere 5 NTU increase (about 20 mg/L) has been shown to reduce the productive area of a lake by about 80 percent (Lloyd et al. 1987). The sensitivity of submersed plants to turbidity can be expressed by the ratio of the depth maxima of species to the Secchi transparency depth, i.e., the "turbidity tolerance index" (Davis and Brinson 1980). Data on depth maxima and ranges for many submersed species are compiled in Davis and Brinson (1980). The more shade-tolerant non-emergent herbaceous species are listed in Table 8.

Vegetation removal. Harvesting of "aquatic weeds" comprises a direct impact on submersed vegetation, and can shift the community composition at least temporarily. Species richness can either increase or decrease, depending on the initial state and species that are harvested (Sheldon 1986). At the deepwater edge of lacustrine wetlands with submersed plants, milfoil (Myriophyllum spicatum) frequently becomes dominant following the catastrophic alteration of more diverse communities by dredging, herbicides, disease, storms, herbivory, or other factors (Nichols 1984).

Removal of woody overstory generally increases herbaceous vegetation biomass and diversity (Madsen and Adams 1989). In the Prairie pothole region, specific information on shifts in community composition as a result of vegetation removal from grazing, haying, and cultivation, is reported by Kantrud et al. (1989: Appendix B). Annual burning, at least of emergent wetlands of the mesic pine-wiregrass savannas of North Carolina, can increase species richness (Walker 1985).

Thermal Alteration. Changes in wetland thermal regime can cause changes in production and shifts in species composition of the herbaceous plant community (Allen and Gorham 1973, Haag and Gorham 1977). An eventual shift from perennial and woody species to annual and herbaceous species may also occur in wetlands exposed to intermediate degrees of thermal warming (Dunn and Scott 1987, Sharitz et al. 1974). Changes are due both to physiological factors and (in northern wetlands) to changes in ice cover (Geis 1984) and growing season length. Most aquatic plants are killed by temperatures warmer than 45°C for 10 minutes, and by somewhat cooler temperatures for longer periods (Christy and Sharitz 1980). Despite this fact, and the fact that macrophyte species richness may be positively correlated with temperature across broad geographic regions, temperature in itself is probably not a major factor governing the distribution of herbaceous wetland plants (Pip 1989).

Changes in community composition as a result of thermal alteration begin with changes in the germinations, growth, and survival of individual species. For example, the introduction over one year of continuously discharged heated water into a Wisconsin marsh resulted in failed shoot emergence, spring emergence instead of fall emergence, fewer number of shoots, and greater height of shoots in the sedge, Carex lacustris (Bedford 1977). Seedling survivorship of one common floodplain species, Ludwigia leptocarpa, was reduced at 42°C. (Christy and Sharitz 1980). Seedling germination of this species did not vary significantly over the range 22-42°C. Cattail, Typha latifolia, was killed as the probable result of heat-induced depletion of non-structural carbohydrates in its underground storage organs. This cattail may grow best at a water temperature of 30°C, but survival is poor at 35°C and seed germination requires temperatures of 13-24°C (Jones et al. 1979). The common reed (Phragmites communis) may grow best when temperatures fluctuate within the 20-30°C range (Haslam 1973), and reed canarygrass (Phalaris) may grow best at about 25°C (McWilliam et al. 1969). For most species, these values vary by genetic population, life stage, duration of exposure, day length, light intensity, and other factors.

Table 8. Examples of Aquatic Plants That May Indicate Reduced Light Penetration Due to Greater Turbidity or Shade.

From Davis and Brinson (1980) and other sources. Note that these species may occur as well in wetlands that are NOT turbid, although usually in smaller proportion relative to other species.

Alisma plantago-aquatica
Ceratophyllum demersum
Eichhornia crassipes
Elodea canadensis
Heteranthera dubia
Hydrilla verticillata
Lemna minor
Myriophyllum spicatum
Najas flexilis
Najas guadalupensis
Najas minor
Nuphar lutes
Potamogeton crispus
Potamogeton pectinatus
Potamogeton perfoliatus var. bupleuroides
Potamogeton pusillus
Potamogeton richardsoni
Riccia fluitans
Ricciocarpus natans
Spirodela polyrhiza
Vallisneria americana
Zannichellia palustris

Dehydration. Deviations of seasonal and annual hydrologic cycles from their "normal" regime (including stabilization of usually fluctuating regimes) can profoundly affect structure of herbaceous wetland plant communities, perhaps even more so than the actual magnitude of the deviation (Hartog et al. 1989, Zimmerman 1988). In some cases, community changes reflect the "intermediate disturbance" hypothesis, wherein "moderate" deviations from "normal" conditions increase community diversity. For example, in

Okefenokee Swamp in Georgia, Greening and Gerritsen (1987) found greater species diversity and variation in biomass at a site where drawdown was occasional and less predictable than at more predictable sites.

Many herbaceous plant communities, particularly those with rigid stems (e.g., cat-tail, common reed) can endure (and may even require) periods of a few hours or days of occasional dehydration without changing. Even a few non-rigid species can survive two or more weeks of exposure, e.g., water milfoil (Myriophyllum spicatum), bladderwort (Utricularia gibba), duckweed (Lemna minor), pondweed (Potamogeton pectinatus), and Ceratophyllum demersum (e.g., Cooke 1980).

However, if dehydrated shorelines subside (collapse) or complete water level drawdown is sustained over many days (particularly if it occurs during the growing season and results in desaturation of sediments) dehydration can trigger significant changes in wetland community structure. This is largely due to the increased availability of nutrients as sediments become desaturated and oxidized, and partly due to enhanced germination of seeds of wetland plants that have lain dormant for years in sediments.

In wetlands that are strongly influenced by ground water discharge, erect vegetation may be less vulnerable to effects of drawdown, because sediments are less likely to become totally dewatered during intentional drawdown (Cooke 1980). Effects are likely to be most severe when drawdown occurs during extremes of heat or cold.

In the short-term, complete drawdown often shifts the balance of community structure in favor of emergent and woody species, and away from submersed species. In the Southeast, aggressive aquatic plants such as alligatorweed (Alternanthera philoxeroides) and naiad (Najas flexilis) can increase following partial drawdown, while muskgrass (Chara vulgaris), water lily (Nuphar spp.), and water hyacinth (Eichhornia crassipes) can decrease (Holcomb and Wegener 1971, Lantz et al. 1964). In prairie potholes, complete water loss year after year results in reduced richness even of herbaceous plants, with Carex and Polygonum generally becoming dominant (Driver 1977). However, partial drawdown, particularly if it occurs for short periods, may greatly increase macrophyte biomass and growth, due to enhanced nutrient and light availability that otherwise limit submersed species (Wegener et al. 1974). In Minnesota peatlands, artificial drainage resulted in increased dominance of the sedge Carex lasiocarpa (Glaser et al. 1981).

In Indiana, woolgrass (Scirpus cyperinus) was believed to indicate dessication and related disturbance of former wetlands (Wilcox et al. 1985), as was the sedge, Carex antherodes, in central Canada (Millar 1973). Woolgrass, along with reed canary-grass (Phalaris) tolerated severe water level drawdown in a New York reservoir (Burt 1988). In temporarily drained wetlands, Mallik and Wein (1986) found that Typha (cat-tail), Calamagrostis canadensis and Brachythecium salebrosum had highest cover values. Cover and stem density of Typha increased after draining, while plant height and stem diameter decreased, compared to a flooded area. Typha may not be a good indicator of wetland dehydration, however, as the same study showed that on the flooded area, Typha, Sphagnum squarrosum (a moss) and Pellia epiphylla had the highest cover values.

In a literature review on the effects of lake drawdown for control of macrophytes in Wisconsin eutrophic lakes, Cooke (1980) surmised that only three species--Brasenia schreberi (a water shield), Hydrochloa carolinensis, and Potamogeton robbinsii (a pondweed) **always** decline following temporary drawdown, and Nuphar spp. and Myriophyllum spp. **often** decline following drawdown. Species that appear always to increase following temporary drawdown include Alternanthera philoxeroides (alligator weed), Lemna minor (duckweed), Leersia oxyzoides (cutgrass), and Najas flexilis.

In southern Florida, drainage of wet prairies and cypress domes results in increased sawgrass, broomsedge (Andropogon), common reed (Phragmites communis), maidencane (Amphicarpum), chainfern (Woodwardia), and many graminoids and shrubs; in deeper waters of wetlands, cattail (Typha) may increase (Alexander and Crook 1974, Rochow 1983, Rochow and Lopez 1984, Worth 1983, Atkins 1981).

A wealth of qualitative information about hydrologic tolerances of plants has been compiled for the U.S. Fish and Wildlife Service's "National List of Plant Species that Occur in Wetlands" (Reed 1988). This publically-available database classifies all U.S. wetland plants according to their fidelity to wet environments, i.e., obligate (nearly always in wetlands) or facultative (usually or sometimes in wetlands). As one might imagine, the obligate taxa **in general** tend to be less tolerant of desiccation than the facultative taxa listed in that database. Information on hydric preferences of species might be numerically summarized using the index of Michener (1983).

The U.S. Fish and Wildlife Service, through the National Ecology Research Center in Fort Collins, Colorado, has also compiled information on "moist soil management techniques." Use of proposed models will allow users to predict the effect of water level changes on herbaceous wetland plants, or perhaps conversely, what the presence of particular plants suggest about prior hydrologic regimes.

Drawdown of wetland water levels in some regions results in increased susceptibility to fires, which in turn can trigger significant changes in wetland chemistry and vegetation.

Inundation/impoundment. Effects of inundation on emergent herbaceous species are extensively compiled in Fredrickson and Taylor (1982), Knighton (1985), and Whitlow and Harris (1979). Herbaceous species of the prairie pothole region are classified according to 12 life history types, related to flooding regime, by van der Valk (1981). A few additional studies that have examined inundation effects on herbaceous plants are summarized here.

Increased water levels in aquatic bed (submersed and floating-leaved plant) wetlands appear to have little effect in some instances (Davis and Brinson 1980). However, water level increases in other instances may result in increased wave action and initially greater turbidity, which is detrimental to many aquatic plants.

Addition of permanent open water to a non-permanently flooded, emergent wetland increases the opportunity for invasion by many submersed and floating-leaved species, and generally results in an increase in on-site species richness. Normally aggressive, perennial emergents such as purple loosestrife, cat-tail, common reed, and water hyacinth may be reduced or eliminated along with less aggressove species as flooding increases.

Although species richness of an entire non-permanently flooded wetland sometimes declines for a few years after flooding becomes permanent (Sjoberg and Danell 1983), overall community richness may change only slightly in the long term, and only the position of the submersed, emergent, and meadow zones may shift (Harris and Marshall 1963, van der Valk and Davis 1976). Such zonal shifts serve as indications of long-term water level change within a wetland (Botts and Cowell 1988). They occur as dormant seeds of wetland plants, which require specific water depths for germination (Moore and Keddy 1988), germinate along the upland boundary as a result of the new flooding. At the same time, down-gradient species subjected to inundation may be lost as a result of suffocation, build-up of compounds toxic to roots, and alteration of physical conditions, e.g., erosion and scour. Floating-mat vegetation may survive.

The effects of flooding also will depend on flooding depth, frequency, duration, dominant plant species, sediment type, water velocity, and other factors. Short periods of flooding (days or weeks) were reported in Wisconsin to have no effect on wetland community composition (e.g., Nichols et al. 1989). Assemblages of herbaceous wetland plants can be a more sensitive indicator of water level change than assemblages of woody wetland plants, which respond more slowly (Paratley and Fahey 1986).

Deviations of seasonal and annual hydrologic cycles from their "normal" regime in wetlands, and particularly, the elimination of seasonal fluctuations, may reduce overall plant species richness. Native and perennial species, particularly grasses and sedges, may be replaced by taxa that are more aggressive, exotic, clonal,

and/or annuals. Depending on the initial water level, these commonly include cat-tails, bulrush, pickerelweed (Sagittaria) and pondweed (Pontederia)(e.g., Botts and Cowell 1988, McIntyre et al. 1988). In some cases, community changes reflect the "intermediate disturbance" hypothesis, wherein "moderate" deviations from "normal" annual hydrologic conditions increase community diversity.
In wetlands along Lake Erie in Ohio, diking of a marsh increased the dominance of liverwort (Riccia spp.), duckweeds (Lemna minor, Spirodela polyrhiza), coontail (Ceratophyllum demersum), water milfoil (Myriophyllum speciatum), pondweeds, and bladderwort (Utricularia vulgaris) (Farney and Bookhout 1982). Permanent inundation of other marshes has decreased the number of plant species and the dominance of Carex spp. (Farney and Bookhout 1982, Sjoberg and Danell 1983).

In Colorado, subalpine wetlands flooded for longer than about 30-45 days during the growing season had fewer emergent plant species than those inundated for shorter periods (Cooper and Emerick 1989). In Sweden, lakeshore wetlands with less than 40-60 days of flooding during the growing season had maximum richness and cover of macrophytes, in contrast to those flooded for longer periods (Nilsson and Keddy 1988).

Cattails generally tolerate deeper water than most rushes (Lathwell et al. 1973), which tolerate deeper water than most sedges (van der Valk and Davis 1976). Cattail (Typha) can dominate wetlands with water depths generally greater than 15 cm for 6 to 12 months (Mall 1969). The common reed (Phragmites) typically occurs in water depths of 0 to 1.5 meters (Haslam 1970, Spence 1982). For most species, these values vary by genetic population, life stage, duration of exposure, water chemistry, and other factors. The horse-tail, Equisetum fluviatile, appeared to be the most tolerant of several emergent species to modest increases in water depth (Sjoberg and Danell 1983). Depth-to-water-table preferences of many peatland species are given by Jeglum (1971).

As noted above, a wealth of qualitative information about hydrologic tolerances of plants is represented by the U.S. Fish and Wildlife Service's "National List of Plant Species that Occur in Wetlands" (Reed 1988), and in their models currently being developed for moist-soil management and in-stream flow management.

Fragmentation of Habitat. We found no explicit information on macrophyte community response to fragmentation of regional wetland resources. Biomass and cover of submersed wetland plants generally decreases with increasing lake size, while the converse is true for emergent species (Duarte et al. 1986). In England, Helliwell (1983) found greater macrophyte species richness in larger wetlands, but a large amount of the variation could be attributed to other factors. Larger wetlands also may have greater macrophyte richness because they tend to be visited more often than smaller wetlands by birds and other animals capable of introducing new plants (Pip 1987). However, regionally rarer species often occur in small wetlands with unique physical and chemical environments (Moore et al. 1989). One can surmise that species with broad environmental tolerances and that disperse easily (e.g., Godwin 1923) might be least affected as wetlands become more isolated from one another.

Other human presence. The bottoms of Sierra lakes in California with higher levels of human visitation had more coverage with rooted macrophytes (Isoetes, Anacharis, Nitella) and bottom algae (Rhizoclonium) than those less frequently visited; this phenomenon was evident even in lakes where use had been restricted for 10 to 20 years (Taylor and Erman 1979). Trampling and other impacts on riparian wetland vegetation are documented in Cole and Marion (1988) and in studies of wetland buffer zones in New Jersey.

In wetlands of "developed" watersheds in New Jersey, uncharacteristic herbaceous species replaced endemic ones, and herbs and vines were more prevalent than in "undeveloped" watersheds (Ehrenfeld 1983, Schneider and Ehrenfeld 1987). Common reed (Phragmites) typically characterizes many disturbed, nutrient-poor wetlands (Haslam 1971), as do woolgrass and soft rush (Juncus effusus) in Pennsylvania (Hepp 1987). In Ohio, increased eutrophication, warming, and turbidity were implicated in the decline of Najas gracillima and N. flexilis, and an increase in N. marina, N. minor, and N. guadalupensis over a 70-year period (Wentz and Stuckey 1971).

Channels with heavy shipping traffic connecting the Great Lakes had less dense beds of submersed macrophytes than in channels with less ship traffic. Dominance shifted from Myriophyllum spicatum, Elodea canadensis and Heteranthera dubia in the relatively undisturbed channel to Characeae, Potamogeton richardsonii, and Najas flexilis in the disturbed channel (Schloesser and Manny 1989). Similar reductions of macrophyte biomass from recreational boating have been found in Europe, as compiled by Liddle and Scorgie (1980) and Murphy and Eaton (1983).

6.2 SAMPLING METHODS AND EQUIPMENT

Factors that could be important to standardize (if possible) among wetlands when monitoring community structure of macrophytes include:

> age of wetland (successional status), light penetration (particularly for submersed species), water or saturation depth, conductivity and baseline chemistry of waters and sediments, current velocity, abundance of herbivores (particularly muskrat, geese, grazing cattle, crayfish), stream order or ratio of discharge to watershed size (riverine wetlands), sediment type, existence of any prior planting programs, and the duration, frequency, and seasonal timing of regular inundation, as well as time elapsed since the last severe inundation, drought, or fire.

References that provide more detailed guidance on sampling herbaceous wetland vegetation include Fredrickson and Reid 1988a, Moore and Chapman 1986, Mueller-Dombois and Ellenberg 1974, Murkin and Murkin 1989, Phillips 1959, Schwoerbel 1970, and Woods 1975. Guidelines for collecting specimens of aquatic plants for preservation are given by Britton and Greeson (1988), and Haynes (1984). References useful in data analysis are listed in Chapter 3.

If wetlands can be sampled only once, mid-growing season is usually the recommended time. However, many plants are apparent and/or identifiable only for a few weeks of the growing season. Thus, if the aim is to quantify community composition accurately, repetitive visits that account for the diverse phenologies of wetland species should be implemented. Ideally, annual visits could be timed to coincide with year-specific weather conditions, rather than calendar date. For example, Grigal (1985), who sampled vegetation over three years, did field work at slightly different times each year. This increased the chances of finding species in flower, making identification easier. In northern bogs, early fall may be a desirable sampling time, e.g., Wilcox (1986) sampled vegetation in a bog in the first week of September because the maximum number of species was identifiable at that time in Indiana. Optimal sampling times vary geographically.

Whenever possible, plants should be identified in the field rather than collected. Trampling of herbaceous vegetation and compaction of saturated soils during even a single site visit can induce community changes detectable in subsequent visits. Thus, field crews should be as small as possible and follow the same path in and out of a wetland. In riverine and lacustrine wetlands, underwater SCUBA transects can be run (Schmid 1965).

Equipment commonly used to destructively sample herbaceous wetland vegetation (especially submersed species) includes dredges, oyster tongs, plant grappling hooks, steel garden rakes, and similar devices (Britton and Greeson 1988). Equipment designed specifically for sampling herbaceous macrophytes is described by Dromgoole and Brown 1976, Macan 1949, Satake (1987), 1977, Wood 1975, and others.

Types of commonly-collected data on herbaceous wetland communities include species per plot and percent cover. Less often, total stem count per m^2, and stem count per species per plot are determined. Stem counts are usually made only of species perceived to be dominant, and may possibly include a few subdominants.

Herbaceous plant community composition is typically quantified using belt transects or replicate quadrats. Transects and quadrats can be used in all wetland types, but may give less reliable data where vegetation is submerged or otherwise difficult to access. Sampling schemes involving transects or quadrats can yield data that is particularly amenable to statistical analysis. The number, size, and spacing of transects and quadrats in a wetland depends primarily on wetland size, shape, internal heterogeneity (e.g., as perceived during an initial reconnaissance visit and/or from aerial photographs), and the statistical power one wishes to have in detecting spatial change in various community metrics. Larger wetlands require more transects or quadrats, usually spaced farther apart, to accurately characterize overall community composition. More linear wetlands (e.g., narrow fringe marshes along lakes) may require more tightly spaced sampling points, as may ecotone areas along transects. Sampling stations along transects are usually situated at even intervals, and quadrats can be placed evenly (e.g., in a grid), randomly, or clustered. Random placement of plots for the purpose of statistically characterizing a wetland is usually prohibitively expensive, due to the extreme spatial variability of most wetlands (Durham et al. 1985). Plots or transect lines are often marked for future relocation.

In most studies of herbaceous wetlands, investigators have located transects or quadrats in a manner that parallels or spans a likely stressor gradient (e.g., parallel to basin gradient, perpendicular to flow, or parallel to flow path of discharge from a chemical outfall). If the stressor is a point source, the transect should be long enough to allow complete definition of gradients in response to the stressor. Thirty meters was not far enough to show distance effects of wastewater disposal in a bog/marsh system studied by Kadlec and Hammer (1980). To avoid problems of treatment effects spilling over into control plots, Loveland and Ungar (1983) used a randomized block design of 0.25-m^2 plots in each of three vegetation zones. Each zone contained five replications of each block; for controls, five plots were randomly spaced in each zone. In another study of artificial enrichment (Duarte and Kalff 1988), plots were not isolated; fertilized plots were 9 m apart and control plots were 3 m from the corresponding treated plots.

Where multiple, non-overlapping gradients are perceived, transects may be located perpendicular to, or at other appropriate angles to, each other. The number of transects and quadrats in particular cover types within the wetland may also be designed to be proportional to the overall coverage of these cover types.

Transects used for herbaceous community monitoring have ranged upwards from about 100 meters in length (depending on wetland size and shape); quadrats have ranged upwards from 0.05 m^2, and may be rectangular, square, or circular. The minimum effective size can be determined statistically or by plotting of initial data, as described in section 3.3. Based on statistical analysis of dozens of published studies of submersed vegetation, Downing and Anderson (1985) suggested it is better to use small quadrats with great replication than large quadrats with little replication, especially where vegetation stands are not dense. However, they suggested cautious interpretation of this recommendation if small quadrats are being placed in dense macrophyte beds. From a study of 18 Canadian lakes, France (1988) determined that at least 21 replicate samples are required to achieve estimates within 20 percent of the mean biomass, using a sampler with an area of 45.6 cm^2. A different number of replicates would probably be required if determination of richness, rather than biomass, was the objective.

Variable-sized plots also can be used, where plot size depends on life form of vegetation present in proximity to each particular point in the wetland (e.g., Mader et al. 1988). Nested frequency quadrats, in which only the number of times a species is present is recorded--have also been used (e.g., Frenkel and Franklin 1987). These have the advantage of easily data collection, objectivity, and no need to relocate plots, but interpretation depends on plot size and shape and spatial distribution of species, and this approach cannot easily be used to quantify spatial patterns, cover, or biomass.

6.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

In general, the parameters most often measured in studies of herbaceous wetlands are "percent cover" and "biomass (standing crop)." Measures of community structure of submersed or emergent aquatic communities have not been uniformly collected from a series of statistically representative wetlands in any region of the country. Thus, it is currently impossible to state what are "normal" levels for descriptors of community structure such as seasonal plant density or species richness, and their temporal and spatial variability, in any type of herbaceous wetland.

Perhaps the closest approximation of a broad-scale effort is that of Duarte et al. (1986). They looked at just one parameter--lacustrine macrophyte biomass--and examined causes of local and regional variability. From their resultant equations, expected ("nominal") levels of biomass of both emergent and submersed macrophytes in lakes might be estimated. Approximate data describing lake area, depth, slope, and a few other simple parameters are needed to run the calculations.

A few, usually localized, studies of inland wetlands have published Shannon diversity index values. For example:

State	type	N	min.value	max.value	citation
SC	Pfo	?	3.58	3.78	Sharitz et al. 1974
MA	Lab	120	1.47	3.71	Burk 1977
GA	P	?	0.83	1.54	Greening & Gerritsen 1987
WA	P	>300	1.48	2.65	Meehan-Martin & Swanson 1988,1989

Species richness has been reported by many studies, but is not always standardized per unit effort or per unit area as it should. Examples include:

State	type	N	min.value	max.value	citation
SC	Pfo	650	$1.2/0.25m^2$ SE=0.4	$9.6/0.25m^2$ SE=0.8	Dunn and Sharitz 1987
MA	Lab	120	$2.3/0.25m^2$ SE=0.21	$8.3/0.25m^2$ SE=0.56	Burk 1977
IA	Pem	28	$3.2/m^2$ SD=0.5	$8.3/m^2$ SD=2.0	van der Valk & Davis 1976
NY	Pem	90	$7.7/m^2$	$12.2/m^2$	Paratley & Fahey 1986
NJ	P	18	26.8/600m SE=2.7	41.4/600m SE=5.7	Schneider & Ehrenfeld 1987

In addition, Ehrenfeld (1983) summarized her data as follows:

Mean species richness per $600m^2$ (N=16):
Disturbed sites = 33.9 ± 2.17; range 17-47
Pristine sites = 27.8 ± 2.24; range 13-44

A similar study by Morgan and Philipp (1986) reported the following values for coefficient of similarity (based on 12 plots per stream, each plot $600m^2$ in area):

between polluted and unpolluted streams = 16%
among polluted streams = 28%
among unpolluted streams = 26%

Many other studies, although not publishing or summarizing in a useful form their statistics on community structure, have compared herbaceous vegetation among wetlands in a region (i.e., spatial variation). Some of the more systematic or extensive quantitative comparisons include:

> Albert et al. 1987, Bunfield and Evans 1982, Canfield et al. 1983, Canfield and Duarte 1988, Duarte et al. 1986, Ehrenfeld 1986, Henebry et al. 1981, Pip 1979, 1987a,b, Sheath et al. 1986, Stewart and Kantrud 1972, and Terry and Tanner 1984.

One of the more geographically extensive ongoing studies is a survey of vegetation in a large number of Great Lakes wetlands in Michigan (Albert et al. 1987). Survey locations are shown in Appendix B. Another extensive and long-term survey of wetland vegetation is being conducted as part of monitoring studies for the ELF military radiocommunications facility (Blake et al. 1987).

A significant number of studies have compared long-term change (but seldom year-to-year variation) in plant community structure in wetlands, in some cases by use of paleoecological techniques. Changes in most cases have not been quantitatively linked with particular stressors. These chronological studies include:

> Baumann et al. 1974, Brown 1987, Bumby 1977, Burk 1977, Burton and King 1983, Harris et al. 1981, Hale and Miller 1978, Kadlec 1979, Niemier and Hubert 1986, Schwintzer and Williams 1974, Southwick and Pine 1975, Stuckey 1971, van der Valk and Davis 1979, and Wentz and Stuckey 1971.

Qualitative data on community structure of inland herbaceous wetlands appears to be most available for Florida, Minnesota-Michigan-Wisconsin, Louisiana, New York, and North Dakota. Apparently the least amounts of such data are for playa wetlands, and for herbaceous wetlands in the Appalachians, southern Great Plains, and Southwest. Information is most available on impacts of hydrologic alteration and nutrients, and least on impacts of partial burial, contaminant toxicity, and habitat fragmentation.

Reasonably complete, qualitative lists of "expected" wetland herbaceous plants are available for most regions through the USFWS's "National List of Wetland Plants" database, and databases of The Nature Conservancy. Quantitative data are generally most available for vascular emergent species, and less common for submersed plants and mosses. Limited qualitative information may also be available by wetland type from the "community profile" publication series of the USFWS (Appendix C).

7.0 WOODED WETLAND VEGETATION

Discussions in this section focus on trees and shrubs that normally characterize wetlands. In many cases, community structure of wood vegetation is less effective as an indicator of short-term anthropogenic stress than is structure of herbaceous plant communities. This is because species composition of wooded wetlands responds slowly to stress, and is suitable mainly as an integrator of conditions occurring over many months and years.

7.1 USE AS INDICATORS

Enrichment/eutrophication. Changes in community composition of wooded wetlands were attributed to increased nutrients in a Michigan wetland exposed to wastewater, by Kadlec and Hammer (1980), but most studies have been too short to detect significant change. Moreover, changes in nutrient concentration are often associated with changes in hydroperiod, and distinguishing the effects of the two can be difficult. Effects of nutrient increases have more often been detected at the level of the individual plant (e.g., growth, foliage and root nutrient concentrations) than at the community level. However, effects at the individual-plant level can often be eventually translated into effects on community composition. Community-level measurements of woody vegetation may be a poorer indicator of eutrophication than are algal or herbaceous plant communities, which respond more quickly.

Organic loading/reduced DO. Existing literature often does not adequately distinguish the effects on woody plants of organic loading/reduced DO, from the effects of nutrients (discussed above) or inundation (discussed below). For example, in one Minnesota wetland experimentally exposed to wastewater, tree mortality could have resulted from hydrologic changes or methodological variation, and so was not attributed specifically to the effluent (Schimpf 1989).

Feedlot effluent entering an Illinois swamp caused increases in species richness, invasion by new species, and changes in species dominance (Pinkowski et al. 1985). Straub (1984) looked for changes in growth rates in isolated swamps with added fertilizer or wastewater, but after five years found none. Increased tree growth has been noted in Florida wetlands exposed to secondarily treated effluent, but untreated effluent appears to be detrimental (Brown and van Peer 1989, Lemlich and Ewel 1984). Florida regulations for treated wastewater discharges to wetlands specify that "the importance value of any of the dominant plant species (excluding some exotics) occupying the canopy or subcanopy shall not be reduced by more than 50 percent at any monitoring station, or 25 percent overall in the wetland." Exceptions may be allowed if changes can be attributed to catastrophic natural events such as hurricanes or fire. Dominant plant species are defined as those that have a total relative importance value of at least 90 percent during the baseline monitoring period (Schwartz 1987).

Contaminant Toxicity. Few if any studies of have been conducted of community-level response of woody vegetation to contaminants in wetlands. Shallow-rooted species are generally believed to be more sensitive to contaminants than deep-rooted species, due to their greater exposure to waterborne contaminants (Sheehan 1984). A four-year study of the response of wetland species to an oil spill in a Massachusetts inland wetland (Burk 1977) reported post-spill absence of red maple (Acer rubrum), and no effect or increase in sugar maple (Acer saccharinum) and wild grape (Vitis labrusca). Additional toxicological information may be available through EPA's PHYTOTOX database (Royce et al. 1984).

Acidification. There are apparently no community-level studies of effects on wooded vegetation specifically in wetlands.

Salinization. In general, woody plants are more sensitive than herbaceous species because they are usually unable to release salts back into the soil, and must therefore rid themselves of it through leaf loss or dying branches. However, we found very little explicit information on woody wetland community response to salinization. An experimental study in Florida demonstrated stress to individual trees from chloride-enriched water (Richardson et al. 1983). Adverse impacts of road salt on forest communities (probably including some wetland species) have been frequently demonstrated. Some tolerance data also may be available from studies of freshwater tidal wetlands. In North Carolina, Brinson et al. (1985) reported reduced tree basal area and density, and greater litterfall, in forested wetlands temporarily exposed to waters of higher salinity.

Sedimentation/burial. Trees (and especially seedlings) are killed when trunks or stems are partially buried or sediment deposition is sufficient to cut off root oxygen exchange (e.g., Eichholz et al. 1979, Kennedy 1970, Harms et al. 1980, Maki et al. 1980). Floodplain trees in Florida were killed by 0.8 m or more of fill, and tree vigor was reduced by only 0.04 to 0.12 m of fill (Clewell and McAninch 1977). Also, where sedimentation is severe, the frequency and duration of inundation may change, causing shifts in community structure (see below). Siltation can also reduce stem height and diameter growth (Kennedy 1970), thus altering competition and ultimately, community structure. Relatively sediment-tolerant species include eastern cottonwood, baldcypress, water tupelo and black willow (Broadfoot 1973).

Where sedimentation creates shoals in rivers or lakes, these sometimes provide additional substrate for establishment or expansion of wooded wetlands. Moderate amounts of sediment may also have a fertilizing effect.

Turbidity/shade; Vegetation removal. Alteration of the canopy within wooded wetlands may be expected to trigger long-term shifts in community composition of woody species, particularly shrub species. Shade tolerances of most woody species are relatively well-known. Logging has an obvious immediate impact on forested wetlands, but the long-term effects on community structure are poorly known and probably dependent upon initial state and the specific silvicultural procedures used. Repeated "high-grading" (i.e., removal of largest trees of the most valuable species) results in high-density, low-biomass stands of shade-tolerant species such as elm, maple, and willow. Grazing also affects community composition, and woody plants differ in their palatability and thus sensitivity to grazing. Usually, evergreens (particularly cedar) and thorny species are less grazed than other deciduous species, but utilization also depends on local availability.

Thermal alteration. Decreases in plant species richness, basal area, and stem density have occurred in South Carolina wooded wetlands as a result of warmed waters (Scott et al. 1985).

Dehydration. Many woody plant communities in wetlands require the absence of surface water, while others require its presence. In the latter case, many of the species comprising such communities can endure brief periods (e.g., a few hours) of occasional drawdown without changing, so long as sediments remain saturated, and many such communities change little despite weeks, months, or years without surface water (e.g., Parker and Schneider 1975). However, in other wetland communities adapted to flooding, if drawdown is sustained over many days (particularly if it occurs during the growing season and results in desaturation of sediments), dehydration can trigger significant changes in soil chemistry and in wooded wetland community structure. This is largely due to the increased availability of nutrients as sediments become desaturated and oxidized, and partly due to enhanced germination of seeds of woody plants that have lain dormant for years in sediments.

In the short-term, complete drawdown of flood-adapted communities often shifts the balance of community structure in favor of woody species, and away from submersed and emergent species. For example, drainage and groundwater withdrawals near wet prairie and cypress wetlands in Florida have resulted in invasion of these areas by willows (especially in burned and logged areas), maidencane, Brazilian pepper, wax myrtle, dahoon holly, gallberry, saltbush, buttonbush, slash pine, red bay, water oak, cabbage palm, and red maple (Alexander and Crook 1974, Carlson 1982, Duever et al. 1979, Lowe et al. 1984, Richardson 1977). In southwestern riparian wetlands, flow regime alteration by dams, and its effect on sedimentation, has resulted in replacement of native riparian woodlands with non-native salt cedar (Tamarix spp.)(Brady 1985, Stevens 1989). The exact successional pattern will depend on initial state and other factors.

Density and species richness of woody species may increase following drainage and/or across a spatial gradient of decreasing inundation duration (e.g., Thibodeau and Nickerson 1985, Maki et al. 1980). From their limited data, Taylor and Davilla (1986) concluded that the effects of river flow diversion (dehydration) on California riparian communities were more distinguishable in the smallest and largest streams (orders 1 and 4) than in streams of intermediate size (orders 2 and 3).

The presence of seasonally elevated water levels can sometimes be inferred by water marks and drift lines on vegetation, presence of adventitious root "knees", signs of current scouring, subsidence and bank collapse, and other secondary features. If these are found in a wetland whose water levels currently remain low throughout the year, then some evidence is provided that dehydration (e.g., by flow diversion) has occurred.

Also, several investigators have sought to compile hydrologic tolerance or preference data into quantitative metrics. For example, the Corps of Engineers has quantified much of the tolerance data for woody plants in its "Flood Tolerance Index" (FTI), which is based on a weighting of cover estimates according to flood tolerance of the species (Theriot and Sanders 1986). A conceptually similar index is described by Wentworth et al. (1988) and tested by Carter et al. (1988). Either index might be tested to determine its potential for use as an indicator of persistant dehydration, i.e., based on the proportion of facultative species found in an area and reflected by the index value.

Inundation/impoundment. Although the existence of many woody wetland communities is absolutely dependent upon inundation, deviations of seasonal and annual hydrologic cycles from their "normal" regime (including stabilization of usually fluctuating regimes) can profoundly affect structure of woody plant communities in wetlands.

Presence of surface water is generally much more detrimental to woody seedling survival than is simple soil saturation (Hosner 1960). Flooding later in the growing season, when seedlings have leafed out, has the potential for greater impacts than earlier floods (e.g., Scott et al. 1985). Also, stagnant, deepwater flooded conditions may be more detrimental than aerated conditions, e.g., where water is shallow and flowing, organic loading is light, and water levels fluctuate according to a natural seasonal pattern (Teskey and Hinckley 1977).

Species richness of woody wetland plants generally decreases with increasing flood duration (Brown and Giese 1988, Klimas et al. 1981). Several instances have been reported where frequent flooding has selectively removed smaller trees and shrubs (e.g., Ehrenfeld 1986, Maki et al. 1980, Noble and Murphy 1975) and may favor emergent vascular plants and mosses (Jeglum 1975). In western riparian areas, shallow-rooted woody species may be more sensitive to flooding than species with deep tap roots (Stevens and Waring 1985). In an eastern floodplain swamp, mortality was lowest in trees greater than 38 cm dbh (diameter) and greatest in trees less than 13 cm dbh (Harms et al. 1980).

As little as 3 days of flooding during the growing season can result in loss of some woody vegetation through suffocation and compounds toxic to roots (Boelter and Close 1974, Harms et al. 1980, Stoeckel 1967, Jeglum 1975, Davis and Humphrys 1977, Keddy 1989, Maki et al. 1980, Southern Forest Experiment Station 1958), or through alteration of physical conditions, e.g., erosion and scour. Although some species survive at least 3 years of continuous flooding (Green 1947), most cannot survive growing-season inundation for more than a year or two (Broadfoot and Williston 1973).

A wealth of other information about hydrologic tolerances of woody plants has been compiled in several reports, e.g.:

Burton 1984, Whitlow and Harris 1979, Hook 1984, Teskey and Hinckley 1977a,b, 1978a,b,c,d, 1980, Walters et al. 1980.

The U.S. Fish and Wildlife Service's "National List of Wetland Plants" and its FORFLO model (Brody and Pendleton 1987) also compiled substantial databases on hydrologic tolerances of plants in the course of their development. The FORFLO model quantitatively predicts wooded wetland community change, given data on expected hydrologic change. The USFWS and others (e.g., Harris et al. 1985, Kondolf et al. 1987) are currently developing methods for relating hydrologic tolerances of woody plants to instream flows. Intensive, site-specific procedures for quantifying the tolerated days, depths, and seasons of flooding in forested wetlands are demonstrated by Grondin and Couillard (1988).

Individual tree growth may also be affected by inundation. Deviations from normal flooding cycles can reduce tree growth (Malecki et al. 1983). However, temporary flooding by rivers may fertilize floodplain trees, increasing growth (Mitsch et al. 1979). In some cases the basal increment can be larger in the remaining trees as compared to unflooded areas. It should not be assumed that basal growth is a good indicator of flooding stress or survival (Franklin and Frenkel 1987).

Fragmentation of habitat. We found no explicit information on forested wetland community response to fragmentation of regional wetland resources. One can surmise that as the distance between wetlands with seed sources becomes greater and dispersal corridors become hydrologically disrupted, species with narrow environmental tolerances and which do not disperse easily might be most affected. This assumption was used by Hanson et al. (1990), who developed a model which predicted that fragmentation will lead to lower woody plant diversity in riparian wetlands. Those authors classified several woody species according to their seed dispersal ability.

Other human presence. In "developed" watersheds, the frequency of characteristic wetland shrub species was reported to be less than in wetlands in "undeveloped" watersheds (Ehrenfeld 1983).

7.2 SAMPLING METHODS AND EQUIPMENT

Natural factors that could be important to standardize (if possible) among wetlands when monitoring anthropogenic effects on community structure of woody plant communities include:

> age of wetland (successional status), water or saturation depth, sediment type, conductivity and baseline chemistry of waters and sediments, current velocity, abundance of herbivores (particularly beaver, grazing cattle), stream order or ratio of discharge to watershed size (riverine wetlands), and the duration, frequency, and seasonal timing of regular inundation, as well as time elapsed (years) since the last severe inundation, drought, windstorm, or fire.

Seasonal timing of woody plant sampling is less critical than is seasonal timing for sampling of herbaceous plants, because most woody plants are present and identifiable throughout the year. Mid-growing season is usually the recommended time, because of the visibility of seedlings and the relative ease in identifying species then. However, access to woody plants in wetlands may be best in winter if ice is present.

The reference texts and choices for protocols described Section 6 for herbaceous plants generally apply to wooded wetlands as well. However, quadrats are usually larger (at least 1 m^2, and often over 10 m^2) and transects may be longer. Percent cover is less often determined where woody plant canopies are larger than 1 m^2 in diameter. Belt transects and line-intercept methods (Canfield 1941, Mueller-Dombois and Ellenberg 1974) are more frequently employed, and dbh (diameter at breast height) of dominant and subdominant stems is commonly measured. Working in large tracts of bottomland hardwood wetland, Durham et al.

(1985) recommended 0.1 ha fixed-area plots for overstory and saplings, with 0.025 ha subplots for sampling shrubs. The State of Florida's regulations for monitoring of discharge of treated wastewater into wooded wetlands specify that quadrat size shall be at least 100 m^2 for canopy vegetation and 50 m^2 for subcanopy vegetation, and that the number of quadrats shall be that number needed to provide 90% certainty of being within 15% of the mean number of species of the population. Additional guidance for sampling streamside wetlands is given by Ohmart and Anderson (1986).

7.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

In general, quantitative community-level data on wooded wetland vegetation have not been uniformly collected from a series of statistically representative wetlands in any region of the country. Thus, it is currently impossible to state what are "normal" levels for parameters such as seasonal plant density, species richness, biomass, or productivity, and their temporal and spatial variability, in any type of wooded wetlands.

Perhaps the closest approximation of such a data set base is the U.S. Forest Service's Forest Inventory and Assessment database (FIA), and Continuous Forest Inventories (CFI). At least in theory, mean density and species richness could be calculated by state for each of the forest types that characteristically occur in wetlands. A large data set describing these metrics also was collected by the U.S. Army Corps of Engineers, along the lower Mississippi River (Klimas 1988), and another was collected by Jensen et al. (1989) in western riparian systems.

Data on another community metric--importance value--were presented in some of the soil-vegetation correlation studies sponsored by the U.S. Fish and Wildlife Service (e.g., Dick-Peddie et al. 1987, Erickson and Leslie 1987, Hubbard et al. 1988, Nachlinger 1988). In addition, examples of studies of multiple forested wetlands across a region, that quantify woody biomass, stem density, or basal area, include the following:

> Dale 1984, Ehrenfeld 1986, Faulkner and Patrick n.d., Jones 1981, Klimas 1988, Osterkamp and Hupp 1984, Reiners 1972, Robertson et al. 1984.

A few published studies have quantified long-term successional changes in community structure of riparian or other wooded wetland communities, sometimes in the engineering context of reconstructing past flood histories. Examples include Malecki et al. 1983, Schwintzer and Williams 1974, and studies cited in Hupp (1988).

Quantitative data on community composition of wooded wetlands appears to be most available for California, the lower Mississippi basin, and Minnesota-Michigan-Wisconsin; and least for New England wooded swamps, Pacific Northwest swamps, and Midwestern riparian systems. Information is most available on impacts of hydrologic alteration, and least on impacts of partial burial, contaminant toxicity, salinization, and habitat fragmentation.

Reasonably complete, qualitative lists of "expected" wetland woody plants are available for most regions through the USFWS's "National List of Plant Species that Occur in Wetlands" (Reed 1988) and databases of The Nature Conservancy. Also, qualitative information may be available by wetland type from the "community profile" publication series of the USFWS (Appendix C).

8.0 WETLAND INVERTEBRATE COMMUNITIES

8.1 USE AS INDICATORS

Discussions under the heading "Invertebrates" here include aquatic insects, freshwater crustaceans (e.g., amphipods, crayfish), aquatic annelids (e.g., worms), zooplankton, and terrestrial insects (e.g., the butterfly, bog elfin, and others listed by Niering 1985) that are found predominantly in wetlands.

Enrichment/eutrophication. Wetland invertebrates respond strongly to trophic condition. Abundance generally increases with increased nutrient concentrations (e.g., Cyr and Downing 1988, Tucker 1958) and species richness may decrease (Wiederholm and Eriksson 1979) or increase (Tucker 1958). Particular species assemblages of invertebrates have commonly been reported to be useful indicators of lake trophic state (Table 9) and may find similar usage in wetlands. These include:

- aquatic worms (Oligochaeta) (Gatter 1986, Milbrink 1978, Lafont 1984, Lauritsen et al. 1985);
- midges (Chironomidae) (Rae 1989, Wiederholm and Eriksson 1979, Winnell and White 1985);
- snails (Gastropoda)(Clarke 1979a); and
- clams (Sphaeriidae)(Clarke 1979b, Klimowicz 1959).

In particular, the ratios of (a) tubificid worms to aquatic insects, (b) the chironomid subfamilies Tanypodinae and/or Chironomini to the subfamily Orthocladiinae), and/or (c) cladocerans to rotifers, have been reported to increase with increasingly eutrophic conditions (Ferrington and Crisp 1989, Gatter 1986, Radwan and Popiolek 1989, Rosenberg et al. 1984). As species shifts occur with increasing eutrophication, chironomid species richness may decline; however, chironomid biomass and/or abundance increase (Ferrington and Crisp 1989, Johnson and McNeil 1988). Indeed, chironomid emergence was recommended as an efficient indicator of secondary production in lakes by Welch et al. (1988).

Organic loading/reduced DO. Excessive organic loading of surface waters, including wetlands, is known to alter community composition (CH2M Hill 1989), usually reduces invertebrate diversity and evenness (e.g., Sedana 1987), and sometimes reduces density and biomass (e.g., Hartland-Rowe and Wright 1975, Pezeshik 1987, Schwartz and Gruendling 1985, USEPA 1983). However, density and biomass of benthic invertebrates in a southern Quebec wastewater wetland was significantly greater than in unexposed wetlands (Belanger and Couture 1988). Density (Sedana 1987) and richness of invertebrates also increased in an Alabama pond after a single episodic addition of manure, but after four weeks richness declined to less than in a control pond (Deutsch 1988). Florida regulations for treated wastewater discharges to wetlands specify that "the Shannon-Weaver diversity index of benthic macroinvertebrates cannot be reduced below 50 percent of background levels as measured using standard techniques."

Under moderate loading, attached algae may increase and consequently, herbivorous mayflies and midges may dominate the community (Jones and Clark 1987). However, if turbidity and hydroperiod conditions allow submersed or floating-leaved aquatic plants (e.g., Lemna) to out-compete algae, other aquatic invertebrates may become dominant.

Table 9. Examples of Aquatic Invertebrates That May Indicate Eutrophic Conditions in Wetlands.

Only Chironomidae are listed. Compiled from the ERAPT database (Dawson and Hellenthal 1986) and the following references: Harvey and McArdle 1986, Rosenberg et al. (1984), Saether (1975), Strange 1976, and Walker et al. 1985, Wiederholm and Eriksson (1979). Note that these species may occur as well in wetlands that are NOT eutrophic, although usually in smaller proportion relative to other species.

Chironomus attenuatus
Chironomus crassicaudatus
Chironomus riparius
Chironomus stigmaterus
Cryptochironomus blarina
Cryptotendipes casuarius
Cryptotendipes emorsus
Dicrotendipes incurvus
Dicrotendipes modestus
Einfeldia natchitocheae
Glyptotendipes barbipes
Glyptotendipes meridionalis
Goeldichironomus holoprasinus
Harnischia boydi
Harnischia galeator
Kiefferulus dux
Leptochironomus nigrovittatus
Pagastiella orophila
Parachironomus directus
Parachironomus monochromus
Parachironomus scheideri
Paralauterborniella elachista
Paralauterborniella subcincta
Pedionomus beckae
Polypedilum digitifer
Polypedilum illinoense
Polypedilum trigonum
Tribelos quadripunctatus

Pseudochironomus richardsoni
Calopsectra dendyi
Calopsectra xantha
Micropsectra dubia
Tanytarsus buckleyi
Tanytarsus recens
Cricotopus bicinctus
Cricotopus sylvestris
Psectrocladius dyari
Coelotanypus concinnus
Coelotanypus tricolor
Ablabesmyia aequifasciata
Ablabesmyia annulata
Ablabesmyia basalis
Ablabesmyia hauberi
Ablabesmyia mallochi
Ablabesmyia ornata
Ablabesmyia peleensis
Ablabesmyia rhamphe
Guttipelopia currani
Labrundinia johannseni
Labrundinia pilosella
Monopelopia boliekae
Procladius bellus
Procladius denticulatus
Tanypus carinatus
Tanypus punctipennis

Chironomus carus
Chironomus plumosus
Chironomus staegeri
Chironomus tentans
Cryptochironomus fulvus
Cryptotendipes darbyi
Dicrotendipes californicus
Dicrotendipes leucoscelis
Dicrotendipes nervosus
Endochironomus nigricans
Glyptotendipes lobiferus
Glyptotendipes paripes
Harnischia amachaerus
Harnischia edwardsi
Harnischia viridulus
Lauterborniella varipennis
Omisus pica
Parachironomus carinatus
Parachironomus hirtalatus
Parachironomus pectinatellae
Parachironomus tenuicaudatus
Paralauterborniella nigrohalteralis
Paratendipes subaequalis
Phaenopsectra profusa
Polypedilum halterale
Polypedilum simulans
Stenochironomus hilaris
Pseudochironomus fulviventris

Calopsectra confusa
Calopsectra neoflavella
Cladotanytarsus viridiventris
Micropsectra nigripila
Tanytarsus quadratus
Parametriocnemus lundbeckii
Cricotopus remus
Nanocladius alternanthera
Clinotanypus pinguis
Coelotanypus scapularis
Psectrotanypus vernalis
Ablabesmyia americana
Ablabesmyia aspera
Ablabesmyia cinctipes
Ablabesmyia illinoensis
Ablabesmyia monilis
Ablabesmyia parajanta
Ablabesmyia philosphagnos
Ablabesmyia tarella
Labrundinia floridana
Labrundinia neopilosella
Labrundinia virescens
Procladius adumbratus
Procladius culiciformis
Procladius riparius
Tanypus grodhausi
Tanypus stellatus

In an isolated Florida cypress swamp dosed with treated wastewater, the following taxa were dominant:

- Nais obtusa
- Psychoda albicans, alternata
- Chironomus riparius
- Polypedilum convictus, flavus

Invertebrates that were absent (but present in untreated swamps nearby) included the following (Brightman 1976):

- Lioplax subcarinata
- Glyptotendipes lobiferous
- Goeldichironomus sp.
- Tanypus stellatus
- Anomalagrion hastatum
- Orthemis ferraginea

In another Florida wetland, McMahan and Davis (1978) detected no impact on terrestrial invertebrate diversity from wastewater additions, despite eutrophicated conditions that resulted. After addition of manure, some Alabama ponds previously dominated by Cladotanytarsus, Clinotanypus, and Procladius became dominated by Dero, Stylaria, and Physa (Deutsch 1988).

In a Vermont wastewater-impacted wetland, caddisflies, clams, snails, water spiders, crustacea, and all aquatic insects except midges were significantly impacted (Schwartz and Gruendling 1985). The impact was due largely to the shading out of submersed plant substrates by algal blooms. Consequently, herbivorous mayflies and midges can begin to dominate such communities (e.g., Jones and Clark 1987). However, if turbidity and hydroperiod conditions allow submersed or floating-leaved aquatic plants (e.g., Lemna) to out-compete algae, other aquatic invertebrates may become dominant.

Even in the absence of human-related wastewater influences, invertebrate communities in wetlands that naturally have low dissolved oxygen are sometimes depauperate compared to those naturally having greater oxygen (e.g., White 1985). Wetland invertebrates that appear to tolerate low oxygen levels (which typify even some undisturbed wetlands) are listed in Table 10. Ratios of tolerant to intolerant species have often been used to indicate ecological status of surface waters, and could be similarly tested for use in wetlands.

Contaminant Toxicity. The availability of vegetation may be particularly important to invertebrates in wetlands having contaminated, persistently anoxic, or highly saline sediments. In such situations, vegetation provides an colonization surface isolated from sediments, where contaminants often are concentrated; richness and abundance of epiphytic and nektonic invertebrate groups may thus remain high in well-vegetated wetlands (McLachlan 1975).

Under more severe exposure to contaminants (e.g., large ambient concentrations of dissolved metals), aquatic invertebrate species richness and density both decline, at least in shallower wetlands (Ferrington et al. 1988, Krueger et al. 1988, Winner et al. 1975). Richness and density can decline even with levels (of phenols and oil-water ratios) not known to be toxic in laboratory studies (Cushman and Goyert 1984).

Shifts in community composition occur as well. Specifically, shifts in structure away from aquatic insects and toward a community dominated by certain oligochaetes (aquatic worms) have been noted in sediments severely contaminated by heavy metals (e.g., Wentsel et al. 1978, Howmiller and Scott 1977, Winner et al. 1980). Areas that are at least moderately contaminated often are dominated by chironomid midges (Winner et al. 1980, Cushman and Goyert 1984, Rosas et al. 1985, Waterhouse and Farrell 1985) and other aquatic invertebrate species whose adults have wings and short life cycles, e.g., water bugs and water beetles (Borthwick 1988, Courtemanch and Gibbs 1979, Gibbs et al. 1981). However, responses to low levels of copper seem to be family- or genus- specific, rather than occuring at the "order" level of taxonomic

Table 10. Examples of Aquatic Invertebrates That Tolerate Low-Oxygen Conditions in Wetlands.

Compiled from the ERAPT database (Dawson and Hellenthal 1986) and its supporting documents (Beck 1977b, Harris et al. 1978). Note that these species may occur as well in wetlands that are NOT anoxic, although usually in smaller proportion relative to other species.

EPHEMEROPTERA (mayflies):

Callibaetis floridanus
Hexagenia limbata
Caenis diminuta

CHIRONOMIDAE (midges):

Chironomus attenuatus
Chironomus crassicaudatus
Chironomus riparius
Chironomus tentans
Cryptochironomus curtilamellatus
Cryptotendipes darbyi
Dicrotendipes fumidus
Dicrotendipes neomodestus
Endochironomus nigricans
Glyptotendipes paripes
Psectrocladius dyari
Goeldichironomus holoprasinus
Harnischia viridulus
Parachironomus monochromus
Paralauterborniella elachista
Paratendipes albimanus
Polypedilum aviceps
Polypedilum halterale
Polypedilum nigritum
Polypedilum ontario
Polypedilum tritum
Pseudochironomus chen
Cladotanytarsus viridiventris
Tanytarsus buckleyi
Cricotopus belkini
Cricotopus politus
Cricotopus sylvestris
Coelotanypus concinnus
Ablabesmyia aequifasciata
Ablabesmyia mallochi
Ablabesmyia rhamphe
Procladius bellus
Tanypus carinatus
Tanypus neopunctipennis su
Tanypus stellatus
Chironomus carus
Chironomus riparius
Chironomus tentans
Endochironomus nigricans
Glyptotendipes lobiferus
Goeldichironomus holoprasinus
Cricotopus remus
Tanypus carinatus
Tanypus stellatus

Chironomus chelonia
Chironomus plumosus
Chironomus stigmaterus
Cryptochironomus blarina
Cryptochironomus fulvus
Dicrotendipes californicus
Dicrotendipes modestus
Dicrotendipes nervosus
Glyptotendipes barbipes
Glyptotendipes lobiferus
Glyptotendipes meridionalis
Harnischia galeator
Kiefferulus dux
Parachironomus tenuicaudatus
Paralauterborniella subcincta
Phaenopsectra profusa
Polypedilum digitifer
Polypedilum illinoense
Polypedilum obtusum
Polypedilum scalaenum
Pseudochironomus aix
Pseudochironomus richardsoni
Micropsectra nigripila
Brillia flavifrons
Cricotopus bicinctus
Cricotopus remus
Psectrocladius dyari
Coelotanypus tricolor
Ablabesmyia aspera
Ablabesmyia monilis
Larsia decolorata
Procladius culiciformis
Tanypus grodhausi
Tanypus punctipennis
Chironomus attenuatus
Chironomus plumosus
Chironomus stigmaterus
Cryptochironomus fulvus
Glyptotendipes barbipes
Glyptotendipes meridionalis
Micropsectra nigripila
Procladius culiciformis
Tanypus punctipennis

classification (Leland et al. 1989). Additions of heavy metals to aquatic ecosystems may increase the ratio of predators to herbivores and detritivores, at least initially (Leland et al. 1989). Nematodes may be particularly sensitive indicators of contaminant toxicity in wetlands that lack surface water; those of the subclass Adenophorea tend to be more sensitive than those of the subclass Secernentea (Bongers 1990, Platt et al. 1984, Zullini and Peretti 1986).

The commonly used herbicide, Atrazine, has been shown to cause shifts in community composition and emergence times of aquatic insects at a concentration of 2 mg/L (Dewey 1986). Other herbicides used in wetlands have been shown to increase the dominance of invertebrates tolerant of low dissolved oxygen, a result related to the large oxygen deficit commonly caused by decay of massive amounts of plants (Scorgie 1980). Also, oil and associated phenols reduced richness, diversity, and total abundance of aquatic insects in one set of wetland experiments (Cushman and Goyert 1984). The midge *Cricotopus bicinctus* and the aquatic worm *Limnodrilus hoffmeisteri* were more prevalent downstream of than upstream from an oil spill (Penrose 1989).

However, some midges (e.g., *Nilotanypus fimbriatus*) are reportedly very sensitive to oil (Rosenberg and Wiens 1976) and pesticides (Hanson 1952). Mayflies (except burrowing species) are particularly sensitive to metals (Leland et al. 1989, Wagerman et al. 1978), oil (Giddings et al. 1984, Cushman and Goyert 1984), and pesticides (Hurlbert et al. 1972, Ali and Stanley 1982, Van Dyk et al. 1975). Amphipods, at least the genera *Gammarus* and *Hyallela*, and the clam shrimp (*Lynceus brachyurus*) appear to be very sensitive to certain pesticides. As indicators of contamination, these freshwater shrimp have the added benefit of being relatively stationary (i.e., because they do not emerge and fly away like aquatic insects, their presence may be more indicative of the longer-term conditions of a wetland). Dosed populations have taken up to a year to recover. They occur in most wetlands with standing water, and their response to pesticides has been documented in prairie pothole wetlands (Borthwick 1988) and Maine bog ponds (Gibbs et al. 1981, Courtemanch and Gibbs 1979). They also have been reported as absent from stormwater treatment wetlands while present in nearby unexposed wetlands (Horner 1988).

It is conceivable that other crustacea, such as crayfish, respond similarly. However, few community-level data are available. Crayfish are damaged by copper levels of greater than 0.5 mg/L (Hobbs and Hall 1974), cadmium levels greater than 10 mg/L (Fennikoh et al. 1978), and mercury levels greater than about 2 mg/L (Doyle et al. 1978).

In wetlands that lack permanent standing water (e.g., bogs, floodplains), data on heavy metal toxicity from terrestrial invertebrate studies may be pertinent. A summary of such studies by Bengtsson and Tranvik (1989) reports the following:

- Species richness and, less often, total abundance of terrestrial invertebrates declines with increasing metal concentration;
- Rare species appear more sensitive than common, widespread species;
- Least sensitive groups include soft-bodied invertebrates such as earthworms, terrestrial herbivores such as ants and weevils, and invertebrates that inhabit the upper soil layers;
- Oribatid mites, the nematode suborder Dorylaimina, and many ground beetles (Carabidae) are highly sensitive, whereas springtails (Collembola) as a whole are less so.

The authors suggest maximum allowable concentrations for lead of less than 100-200 mg/kg; less than 100 mg/kg for copper; less than 500 mg/kg for zinc, and less than 10-50 mg/kg for cadmium.

Other thresholds of invertebrate toxicity for metals and/or synthetic organics are given by Johnson and Finley (1980), USEPA (1986), EPA's "AQUIRE" database and the US Fish and Wildlife Service's "Contaminant Hazard Reviews" series that summarizes data on arsenic, cadmium, chromium, lead, mercury, selenium, mirex, carbofuran, taxaphene, PCBs, and chlorpyrifos.

Although not directly manifested in changes in community structure, physical deformities of individuals often accompany severe pollution. Midges with deformed mouth parts were noted in areas of synthetic-coal-derived oil pollution (Cushman and Goyert 1984).

Acidification. Knowledge of acidification effects on wetland invertebrate communities comes mainly from studies in acidified lakes and streams exposed to mine drainage. As compared to circumneutral or slightly alkaline waters, acidic waters (natural or recently induced acidity) generally have less invertebrate biomass and/or species richness, lower ratio of consumers to producers, and fewer clearly dominant taxa (e.g., Friday 1987, Hall and Likens 1980, Harvey and McArdle 1986, Letterman and Mitsch 1978, Parsons 1968, Smock et al. 1981, 1985, Thorp et al. 1985, Walker et al. 1985a, Warner 1971). However, several studies, e.g., those of some acidic "blackwater" streams, have detected no significant differences in lake or stream invertebrate numbers or richness attributable to pH differences (e.g., Bradt et al. 1986, Bradt and Bert 1987, Collins et al. 1981, Crisman et al. 1980, Kelso et al. 1982, Winterbourn and Collier 1987). The effects of acidification may interact with and possibly be overshadowed by trophic conditions of wetlands (Brett 1989, Kerekes et al. 1984, Schell and Kerekes 1989).

Shifts in community composition are probably the most frequently measured effect of acidification. Particularly acid-sensitive are species of gastropods (snails), pelecypods (clams and mussels), daphnids, ephemeropterans (mayflies), amphipods (freshwater shrimp), and some midges (particularly the subfamilies Chironominae and Orthocladinae) (Allard and Moreau 1987, Bell 1971, Friday 1987, Hall et al. 1980, Harvey and McArdle 1986). Some of the first species to be affected by acidification are crustacea-- the predaceous copepod, Epischura lacustris (Sprules 1975), and the freshwater shrimp, Hyalella azteca (Zischke et al. 1983) and Gammarus lacustris. Taxa reported to be more prevalent under acidic conditions include oligochaetes, acarids (water mites), the phantom midge, Chaoborus, and midges of the subfamilies Tanypodinae and possibly Chironomini (Allard and Moreau 1987, Bradt and Bert 1987). A few caddisflies, freshwater sponges, dragonflies, water bugs (Corixidae), water beetles (Dytiscidae), and Tanytarsini midges tolerate weakly acid conditions (Fowler et al. 1985, Walker et al. 1985a). Species of midges and caddisflies known to occur under acidic conditions are listed in Table 11, based on data compiled by Beck (1977b) and others.

Salinization. Naturally saline, nontidal wetlands typically have low diversity of aquatic invertebrates (Kantrud 1989) and are dominated by brine shrimp (Artemia), brine flies (Ephydra), and a few species of midges and aquatic worms. Severe increases in salinity of freshwater habitats also can diminish invertebrate community biomass and species richness. However, rather few data have been collected specifically from inland brackish wetlands, so relative tolerances of species to increased salinity are poorly known.

Other taxa known to be relatively tolerant include certain species of midges, mosquitoes, aquatic worms, dragonflies, water bugs, and water beetles (Kreis and Johnson 1968). Crayfish generally require salinities less than 15 ppt (Loyacano 1967). Former salt marshes that were converted to freshwater wetlands were found to have fewer midges of the subfamily Orthocladiinae than expected (Walker et al. 1985a).

If more information were available on tolerances, such data might be used (e.g., as a ratio of salt-tolerant to salt-intolerant species) in conjunction with background chemical data to indicate stress to wetlands from irrigation runoff water, cultivation of saline soils, coastal saltwater intrusion, or other salinity sources.

Table 11. Examples of Invertebrates That May Tolerate or Prefer Acidic Conditions in Wetlands.

Compiled from the ERAPT database (Dawson and Hellenthal 1986) and the following references: Beck 1977b, Kimerle and Enns 1968, Smock et al. 1981, Walker et al 1985a. Note that these species may occur as well in wetlands that are NOT acidic, although usually in smaller proportion relative to other species.

Ablabesmyia americana, *A. aspera*, *A. basilis*, *A. hauberi*, and *A. parajanta*, *A. peleensis*, *A. philosphagnos*
Chironomus (some species)
Cladopelma (some species)
Cladotanytarsus (some species)
Corynoneura taris
Cryptotendipes casuarius
Dicrotendipes incurvus, *D. leucoscelis*
Guttipelopia currani
Harnischia amachaerus, *H. boydi*
Krenosmittia (some species)
Labrundinia floridana, *L. johannseni*, *L. neopilosella*, *L. virescens*
Lauterborniella varipennis
Metriocnemus abdomino-flavatus, *M. hamatus*, *M. knabi*
Monopelopia tillandsia
Monopsectrocladius (some species)
Nilotanypus americanus
Nimbocera (some species)
Omisus pica
Orthocladius annectens
Pagastiella orophila
Parachironomus alatus, *P. scheideri*
Paramerina anomala
Polypedilum braseniae, *P. nymphaeorum*, *P. obtusum*
Procladius bellus
Tanypus neopunctipennis
Tanytarsus (some species)
Thienemannimyia senata
Tribelos quadripunctatus
Trissocladius (some species)
Dugesia tigrina
Nais (some species)
Limnodrilus hoffmeisteri
Aulodrilus piqueti
Crangonyx (some species)
Hydracarina
Callibaetis diminuta
Caenis diminuta
Oxyethira (some species)
Palpomyia (some species)

Burial/sedimentation. High rates of sedimentation (7 cm/yr) resulted in lower diversity, richness, and total community biomass in a southern river system (Cooper 1987). Fine-particle sediments, particularly if anoxic, support reduced diversity and richness of invertebrates (Wilbur 1974). Species of mayflies and chironomids Species of mayflies and chironomids that feed mainly on algae are particularly affected, while burrowing invertebrates might be expected to be least-affected. In Lake Erie, the abundance of tubificid worms was correlated with the sediment accumulation rate and organic carbon flux (but not to organic carbon) (Robbins et al. 1989). Excessive sedimentation may be indicated by absence of the freshwater bryozoans, e.g., Pectinella magnifica, and the fingernail clam Sphaerium rhomboideum (Cooper 1987, Cooper and Burris 1984).

Turbidity/Shade; Vegetation Removal. Removal of aquatic bed vegetation can increase algae in wetlands, thus increasing the ratio of herbivorous species (e.g., certain mayflies) to detritivorous species (e.g., certain midges and worms). Submersed plants and logs have among the highest densities and species richness of any aquatic substrate, e.g.:

> Armstrong and Nudds 1985, Boerger et al. 1982, Chubb and Liston 1986, Crowder and Cooper 1982, Durocher et al. 1984, Dvorak and Best 1982, Floyd et al. 1984, Gilinsky 1984, Hall and Werner 1977, Kallemeyn and Novotny 1977; Kimble and Wesche 196, Krecker 1939, Krull 1970, Menzie 1980, Minkley 1963, Miller et al. 1989, Mittelbach 1981, Poe et al. 1986; Scheffer et al. 1984, Schramm et al. 1987, Teels et al. 1978, Voigts 1976, Ware and Gasaway 1978, Wetzel 1975.

Indeed, equations for predicting the density of aquatic invertebrates in submersed vegetation (lacustrine aquatic bed) have been developed by Cyr and Downing (1988), using data on biomass of individual macrophyte species and season. Thus, removal or loss of aquatic vegetation due to shading/turbidity can be expected to profoundly affect the invertebrate resource (e.g., Bettoli 1987, Vander Zouwen 1983). On the other hand, selective removal of dense macrophyte stands can increase density, biomass, and/or richness of remaining invertebrate communities (e.g., Beck et al. 1987, Broschart and Linder 1986, Kaminski and Prince 1981, Kenow and Rusch 1989, Murkin and Kadlec 1986). Removal of the canopy of one forested floodplain wetland had little effect on aquatic invertebrate richness and density (Boschung and O'Neil 1981). The degree to which vegetation removal has a neutral or beneficial effect on macroinvertebrates may depend partly on the type of removal procedure (e.g., mechanical thinning, ditching, burning, herbicides, crayfish introduction) and the spatial patterns created (Nelson and Kadlec 1984).

Non-aquatic invertebrates may also respond to removal of woody and emergent vegetation. For example, a decline of wetland spider richness accompanied peat harvesting in a bog (Koponen 1979).

Thermal Alteration. Heated effluents generally reduce the richness of invertebrate communities in wetlands and may either increase or decrease their density and productivity (Gibbons and Sharitz 1974, McKnaught and Fenlon 1972, Nichols 1981, Poff and Matthews 1986, Whitehouse 1971, Wiederholm 1971). Increases in secondary productivity are the result of higher primary productivity associated with warmer temperatures and longer growing seasons. Crayfish generally cannot tolerate temperatures greater than about 30 C (Becker et al. 1975). Backswimmers (Corixidae) and midges appear to tolerate moderately warmed surface waters (Gibbons and Sharitz 1974). Temperatures of over 40 C apparently do not significantly affect the life cycle of the midges Chironomus sp., Tanypus neopunctipennis, or Tanytarsus sp.; where deep, soft substrates are available as refugia for burrowing species, damage from thermal increases may be lessened (Coler and Kondratieff 1989). The ratio of burrowing oligochaetes, nematodes, gastropods, chironomid midges, and nektonic invertebrates to other aquatic invertebrates might thus be tested as one indicator of thermal disturbance.

Dehydration, Inundation. Water levels profoundly affect the abundance and community composition of invertebrates (Reid 1985, Wiggins et al. 1980). Addition of permanent open water to a non-permanently flooded wetland increases the opportunity for invasion by many submersed and floating-leaved species that provide complex substrates for aquatic invertebrates. This consequently can result in an increase in on-site species richness, and perhaps increased density, of wetland invertebrates. For example, inundation of emergent wetlands was noted to increase the density, biomass, and richness of invertebrates (Huener 1984), and cause a shift in community composition toward herbivores and detritivores (Murkin and Kadlec 1986). For Mississippi River borrow pit wetlands, "days flooded" was the most significant factor explaining invertebrate density in a multivariate regression; flooding in the sampled wetlands ranged from 24 to 115 days annually, with a mean of 81 (Cobb et al. 1984).

However, if inundation in some wetlands is prolonged (throughout the growing season) and deep, the resulting oxygen and light deficits may result in diminished richness and density of aquatic plants (Ebert and Balko 1987). Prolonged growing-season flooding, when it occurs in wetlands that have no prior history of such flooding, results in diminished invertebrate density and richness (Driver 1977, Hynes and Yadev 1985, Neckles et al. 1990). In forested floodplain wetlands, invertebrate species richness and abundance decrease with increasing soil moisture and flood frequency (e.g., Uetz et al. 1979) and with disruption of normal sequencing of flooding (Sklar and Conner 1979).

When wetlands that normally contain standing water are almost totally dehydrated for short periods (i.e., "drawdown"), the result is usually a major increase in nutrients, algae, and invertebrate density (Benson and Hudson 1975, Reid 1985, Wegener et al. 1974). This effect may be less pronounced if a dense canopy prevents sufficient light for algal growth, and exchange rates of wetland water with adjacent waters are minimal. Also, less mobile taxa, such as freshwater clams, may be particularly sensitive to drawdown. They can become stranded and perish during rapid drawdown unless underlying sediments remain saturated and soft so individuals can burrow down into the saturated zone (Jiffry 1984).

Although invertebrate density may increase following reflooding of dehydrated wetlands, invertebrate richness may not, particularly if sediments have become heavily oxidized and hardened during exposure (Hunt and Jones 1972). If wetlands are dehydrated irregularly and rapidly (e.g., by frequent passage of large ships) or for long periods (e.g., reservoir fluctuations), both abundance and richness of invertebrates can decline (Hale and Baynes 1983, Smith et al. 1987).

Invertebrate taxa can be classified into groups (response guilds) related to their life cycles and preference for particular wetland hydroperiods. Conceivably, ratios of these groups (e.g., density-weighted ratio of short-lived/mobile species to longer-lived/immobile species) could be tested as an indicator of wetland hydrologic status, as has been done with midges (Driver 1977) and water beetles (Hanson and Swanson 1989). In prairie pothole wetlands, chironomid diversity was also found to increase with permanency of the hydroperiod (Driver 1977), although contrary evidence is presented by Neckles et al. (1990). Individual taxa might be assigned to the following response groups (Delucchi 1987, Jeffries 1989, McLachlan 1970, 1975, 1985, Wiggins et al. 1980):

- Overwintering Residents: disperse passively; include many snails, mollusks, amphipods, worms, leeches, crayfish.
- Overwintering Spring Recruits: reproduction depends on water availability; include most midges, some beetles.
- Overwintering Summer Recruits: reproduce independent of surface water availability, requiring only saturated sediment; include dragonflies, mosquitoes, phantom midges.
- Non-wintering Spring Migrants: mostly require surface water for overwintering, adults leave temporary water before it disappears in spring or summer; includes most water bugs, some water beetles.

Thus, changes in density-weighted ratios of response groups, monitored from a large regional set of wetlands, might be used to indicate changing hydrologic conditions over time. However, additional research may be needed because some recent evidence suggests that certain taxa (species of Dytiscidae, Corixidae, Ceratopogonidae, Ephydridae, and even Chironomidae) may be unaffected by water regime in some situations (Neckles et al. 1990).

Fragmentation of Habitat. We found no explicit information on wetland invertebrate community response to fragmentation of regional wetland resources. A study of prairie potholes indicated increased diversity with increased wetland size, and the author suggested that might be due to the increased distance of smaller areas from larger and more stable wetlands (Driver 1977). Increased richness and interspersion of plant forms within a wetland can result in increased macroinvertebrate richness and numbers (Voigts 1976).

One can surmise that as the distance between wetlands with colonizers becomes greater, species with narrow environmental tolerances and which do not disperse easily might be most affected. Indeed, in a study of essentially identical wetlands, Jeffries (1989) found that statistical clusters of invertebrate taxa were defined by the distance and surface water connection of their associated wetland from a much larger regional water body. However, even apparently "immobile" species such as amphipods and clams have some capability for dispersal (Swanson 1984).

Landscapes where wetlands are interspersed with uplands can have almost 70 percent more invertebrate species than those containing only uplands (Coulson and Butterfield 1985). In lakes, the species richness of mollusks (Aho 1978, Lassen 1975), midges (Driver 1977), and crustaceans (Fryer 1985) increases with increasing lake area.

8.2 SAMPLING METHODS AND EQUIPMENT

Natural factors that could be important to measure and (if possible) standardize among wetlands when monitoring anthropogenic effects on community structure of invertebrates include:

> age of wetland (successional status), water or saturation depth, conductivity and baseline chemistry of waters and sediments (especially pH, alkalinity or calcium, and organic carbon), sediment type, current velocity, presence of fish, stream order or ratio of discharge to watershed size (in riverine wetlands), density, type, and form of vegetation and woody debris (particularly, total surface area), ratio of open water to vegetated wetland, and the duration, frequency, and seasonal timing of regular inundation, as well as time elapsed since the last severe inundation or drought.

Sampling methods for wetland or lake littoral invertebrates are described in Downing and Rigler 1984, Edmondson and Winberg 1971, Fredrickson and Reid 1988b, Isom 1986, Murkin and Murkin 1989, Witter and Croson 1976, and others. Although addressing streams, the book by Elliott (1971) is an important reference for sampling program design and data analysis.

Larval aquatic invertebrates can be found in wetlands throughout the year. If wetlands can be sampled only once, then the late wet season or beginning of the dry season, if they coincide with the growing season, are usually the recommended time, as density and richness tend to be greatest then (Marchant 1982). Alternatively, if conditions among a series of years are to be compared and the primary desire is to minimize variability, then dry-season measurements made just before the onset of flooding may be best (McElravy et al. 1989). However, the chronology of density peaks can vary even among wetlands in close proximity, possibly due in some cases to differences in predation (Campbell 1983).

In either case, and particularly in disturbed and intermittently flooded wetlands, caution is needed to schedule sampling to coincide with phenologies of particular taxa (Sklar 1985). For example, one might want to avoid sampling immediately after a synchronous emergence of the usually dominant species. Maximum information is often obtained when most invertebrates are within a size range (later instars) retained by nets used to sample them, and can be identified with greatest confidence. For biomass estimates, Hanson et al. (1989) reported that samples collected at 4- and 6- week intervals were very similar to those based on 9 biweekly collections. For a bog stream monitored over 23 months, Boerger et al. (1982) reported a 17-fold variation in midge densities, and even greater variation was reported by Gatter (1986).

The choice of equipment depends largely on the wetland microhabitat to be sampled. Different assemblages of wetland invertebrates inhabit sediments (benthos), rooted plants or algae (phytomacrofauna), open water (nekton), and the surface film (neuston). Subsequent data analysis can use groupings based on ecological niches associated with each taxon (e.g., Cummins and Wilzbach 1985).

A significant problem in analyzing wetland invertebrate data arises from difficulties in determining the spatial dimensions of the area from which a sample was drawn. Accurate estimates of density (individuals per unit area) are difficult to achieve due to difficulties in accurately measuring the complex wetland substrate (submerged plants, tree trunks, emergent plant stems, logs, etc.). To address this, some investigators have removed the substrate along with the collected sample, weighted both, and reported density as weight or number of organisms per unit weight of substrate. In some cases regression coefficients have been calculated to convert plant weights to plant area, which may be further converted to invertebrate density (Downing 1986). Another approach has been to base comparisons among similar wetland habitats on similarity indices and richness (per number of individuals), rather than on density and biomass.

If the objective is to sample invertebrate communities attached to wetland plants (e.g., snails, many mayflies) and the water column, **sweep nets** (dip nets) are commonly used. These are the familiar long-handled insect nets. They may be used in water or air, so long as vegetation is not dense. Usually, they are either swept through a standard length of vegetation, or placed on the bottom and hauled vertically through the water column in a rapid stroke. They are convenient to use, and are particularly suited for capturing large (e.g., crayfish) or quick-moving species not collected by other methods, such as adult dragonflies and water striders. Disadvantages include user variability and the fact that their samples are not strictly quantitative, since the unit of area swept is difficult to accurately determine (Adamus 1984, Plafkin et al. 1989).

Trials by Furse et al. (1981) and Friday (1987) indicate that at least 80 percent of the species found to be present in a particular aquatic plant bed using 5 to 10 sweeps can be captured in half that number. In trial comparisons against a modified Gerking sampler (see below), Kaminski and Murkin (1981) found sweep nets to be just as effective in sampling water-column taxa, although Gillespie and Brown (1966) had come to the opposite conclusion. In wetland studies, sweep nets have been used by Borthwick (1988), Courtemanch and Gibbs (1979), Smith et al. 1987, Voigts 1976, White 1985, and others.

Another option for sampling plant-dwelling invertebrates in wetlands involves directly **clipping the vegetation** and returning it in an enclosed box to the lab. This can be used for both submersed and emergent plants, and provides more precise quantification than does use of sweep nets. Vacuum suction can also be used to remove small invertebrates from foliage in the field (Southwood 1981). Downing and Cyr (1985) found the most cost-effective quadrat size for clipping to be 500 cm^2. Plants were enclosed in a 6-liter plastic box. Clipping aquatic macrophtyes in quadrats of varying sizes yielded five times higher populations than did sampling with Gerking, Macan, Minto, or KUG samplers. Gates et al. (1987) described a sampler useful for taking simultaneous samples of sediment invertebrates and plant-dwelling invertebrates. They found this to give results for plant invertebrates at least as precise and sometimes more accurate than obtained by clipping macrophytes.

A third option for sampling invertebrates of wetland plants involves use of **artificial substrates**. Plants are not sampled directly, but rather, plastic plants or other sterile surfaces (e.g., Hester-Dendy plate samplers) are totally submersed in the wetland water column and allowed to be colonized over a period of at least a month (Macan and Kitching 1972). Because they standardize surface area and texture, collections from substrate samplers are highly comparable to each other, making them attractive for use in monitoring of water column water quality. They also are lightweight, can be used in areas difficult to sample by other means (e.g., deep rivers), and sample processing is relatively clean. However, disadvantages include the fact that a return trip to the wetland is required, vandalism may be a problem, their use is limited to wetlands with surface water, they sample only epiphytic species, and representativeness can be questioned (Adamus 1984).

In an aquatic bed wetland, Gerrish and Bristow (1979) used plastic mimics of the pondweed, Potamogeton richardsonii, interspersed among live experimental plants. Although this yielded no significantly different numbers of invertebrates or species per unit of surface area than were found on real plants, aquatic worms were significantly more common on the artificial substrates, and the substrates did not accurately reflect the densities of invertebrates on the nearby Myriophyllum or Vallisneria plants.

Natural substrates initially devoid of organisms can also be used as colonization substrates. For example, plant litter was placed in boxes made of hardware cloth by Batema et al. (1985) and White (1985), for sampling macroinvertebrates in eastern floodplain forests. Artificial substrates are often ineffective for collecting large crustaceans (e.g., crayfish) and mollusks.

If the objective is to sample invertebrate communities inhabiting wetland sediments, then **dredges**-- also called grab samplers (Ekman, Ponar, etc.)--are often used. They essentially consist of a box with jaws that is lowered onto the sediment. The jaws enclose a specified area of bottom, and retrieve sediments and associated organisms to a sediment depth of about 5 cm. Dredges are used only where surface waters of at least 0.5 m in depth are present, and are not effective where there are rocks, aquatic plants, or logs to jam the jaws. They have been used in wetlands by Bradt and Bert (1987), Driver (1977), and Krull (1970). Estimates of density are only crudely quantitative because jaws seldom close tightly, allowing organisms to escape. Large organisms (e.g., crayfish), water column organisms, and fast-moving species are poorly sampled.

Another option for sampling sediments is to use **core samplers**. Unlike grabs, corers do not have jaws, and instead rely on compactive force or suction to retrieve sediments They suffer the same disadvantages as dredges. Samples may be more precisely quantitative, but the mean size of organisms effectively captured may be smaller, due to the narrowness of corers. Core samplers may be the only option for quantitatively sampling sediment organisms in wetlands that lack surface water, and a variety of designs are available (e.g., Bay and Caton 1969, Coler and Haynes 1966). Core samplers are widely used in paleoecological studies of wetlands. Florida regulations for monitoring treated wastewater discharges to forested wetlands specify that if a core sampler is used, devices with minimum sampling area of 45 cm^2 be used, and that the number of samples at a given station within the wetland be that number needed to be 90% certain of being within 15% of the mean diversity of the population. Where aquatic plants interfere, some investigators have suggested a saw blade might be welded to the leading edge of the corer, for clipping heavy roots and stems (e.g., Murkin and Kadlec 1986). Where sediments are frozen, metal ice spades have been used to collect samples (e.g., Jacobi 1978).
Where sediments or soils are not covered by water (e.g., in peat bogs), pitfall traps and soil extraction techniques can also be used to augment vacuum sampling and sweep-net sampling, and may produce the highest densities and species richness (Coulson and Butterfield 1985).

If the objective is to sample invertebrates that inhabit the water column, **tubular samplers** (e.g., "Gerking samplers", "stovepipes", "Hess samplers", "box samplers") can be used. These are wide cylinders that enclose a standard area of bottom and usually are not designed for effectively penetrating the sediment. In some, the bottom can be sealed off with a sliding door, plug, or similar feature once the sampler is in place. Some have been fitted with a reinforced cutting edge on the bottom. Designs are described by Freeman et al. (1984), Gerking (1957), Korinkova (1971), Hiley et al. 1981, Legner et al. 1975, Mackay and Quadri

(1971), Martin and Shireman (1976), Minto (1977), and Swanson 1978. They are not effective for catching quick-moving organisms, burrowing species, very large taxa, many epiphytic species, or for use in flowing water.

Emergence traps and funnel traps consist of nets or funnels anchored at and just above the water surface. They passively collect aquatic insects as they pass into their winged adult stage and emerge from the water column. Funnel traps are used to collect swimming, air-breathing insects as well as emerging species (e.g., Greenstone 1979, Henrickson and Oscarson 1978, Kaminski and Murkin 1981). Traps--either submerged, at the water surface, or above it-- can be fitted with lights to increase their attraction to some adult insects, for example:

> Aiken 1979, Apperson and Yows 1976, Carlson 1971, Carlson 1972, Espinosa and Clark 1972, Hungerford et al. 1955, Husbands 1967).

A variety of designs for emergence and funnel traps have been tried, for example:

> Corbet 1965, Daniel et al. 1985, Deonier 1972, Lammers 1977, Lemke and Mattson 1969, McCauley 1976a,b, Pritchard and Scholefield 1980, Rosenberg et al. 1980, Voigts 1973, and Washino and Hokama 1968, and some were evaluated by Kimerle and Anderson (1967).

Use of funnel and emergence traps is limited to wetlands containing open patches of surface water during the growing season, when most insects emerge. They can be used in both still-water and slow-flowing wetlands, particularly those difficult to sample by other means, and samples are relatively debris-free and easy to process. Because they are left in place (sometimes for many weeks), they avoid the problem encountered by other samplers of missing key species due to inappropriate time of visit.

Because emerging insects come from a variety of microenvironments, emergence traps can integrate well the extreme spatial heterogeneity within many wetlands. On the other hand, this makes it impossible to standardize or determine the unit of area measured. Thus, they would not be suitable for tracing the leading edge of an effluent plume within a small wetland. Samplers designed for passively collecting terrestrial insects (e.g., light traps, pitfall traps, malaise traps) encounter the same problem. Also, many of the wetland invertebrates most sensitive to pollution do not emerge (e.g., amphipods, aquatic worms, snails), so are not collected by emergence traps. Initial purchase of traps can be costly, and vandalism may be problematic.

In prairie potholes, conical emergence traps were situated at 3 m intervals perpendicular to shore (Driver 1977). To detect immediate effects of pesticide application, Gibbs et al. (1981) emptied emergence traps every two hours, from 6 AM to 8 PM. Normally, traps are left in place for many days or weeks. Welch et al. (1988) used submerged funnel traps to catch emerging midges in a lake. They found no difference in total catch between 0.142-m^2 and 0.283-m^2 trap sizes. Traps with inverted funnels inserted in the jar necks caught more pupae than traps without funnels, and total catch in the traps without jars was 58 percent of the catch in traps with funnels. Rosenberg et al. (1984) submerged their funnel traps, situating them at depths of 1, 2, 3.5, and 4.5 m.

8.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

In general, quantitative data on wetland macroinvertebrates has not been uniformly collected from a series of statistically representative wetlands in any region of the country. Thus, it is currently impossible to state what are "normal" levels for parameters such as seasonal invertebrate density, species richness, or biomass, and their temporal and spatial variability, in any type of wetland.

Perhaps the closest approximation of such a data set is that of Giese et al. (1987). These invertebrate data were collected in part from streams flowing through relatively pristine floodplain wetlands, and thus help serve as a regional baseline for bottomland hardwood wetlands. Collecting a single, timed (30-minute), series of dip-net samples from each site, the investigators found an average of 50-60 invertebrate taxa, containing an average total of about 800 individuals. The Shannon diversity index averaged 4.17 to 4.67 in these relatively pristine lowland streams.

Also, the U.S. Geological Survey is presently initiating a program (NAWQA) to monitor stream invertebrate communities inhabiting a carefully selected sample of watersheds throughout the United States. Although wetlands will not be a specific target of the monitoring, the spatial and regional variability of invertebrates may become better known from this probability sampling approach. Another regionally extensive project was undertaken by Corkum (1989) in the Pacific Northwest/Alaska, and resulted in an ecologically-based classification of stream types for that region. Other data on wetland macroinvertebrates is selectively summarized in Table 12.

Coefficients of variability for invertebrates in streams range from about 0.2 to 0.8 (Eberhardt 1978). Few such values apparently have been published for wetlands. Variation in invertebrate density among habitats within wetlands has been documented in some cases, for example:

> Beck 1977a, Erman and Erman 1975, Gatter 1986, Kansas Biological Survey 1987, Neuswanger et al. 1982, Thorp et al. 1985.

However, only a few studies in the U.S. have quantified invertebrate community differences among a series of wetlands in a region, and have mostly focused on lacustrine or riverine wetlands. These include:

> Bradt et al. 1986, Campbell 1983, Cobb et al. 1984, Erman and Erman 1975, Ferren and Pritchett 1988, Garono and MacLean 1988, Haack et al. 1989, Hepp 1987, Kallemeyn and Novotny 1977, Krull 1970, Lowery et al. 1987, Mathis et al. 1981, McAuley and Longcore 1988, Stoddard 1987, Teels et al. 1978.

Although data exist that quantify year-to-year variation in invertebrate community structure in other surface waters (e.g., McElravy et al. 1989), such studies have apparently not been published for wetlands. Conceivably such unpublished data may be available from sites of the U.S. Department of Energy's National Environmental Research Park system, as well as the following sites of the National Science Foundation's Long Term Ecological Research (LTER) program (that contain studied wetlands): Illinois Pool 19 site, Illinois-Mississippi Rivers sites, New Hampshire Hubbard Brook riparian forest, Oregon Andrews Experimental Forest riparian forest, and Michigan Kellogg Biological Station site.

Quantitative published data on composition of aquatic invertebrate communities appears to be most available for submersed vegetation (aquatic bed wetlands), particularly in the Southeast and Prairie pothole region. Apparently such data are most limited for wetlands that are saturated but mostly lack standing water (e.g., bogs), as well as for playas and non-Southeastern riparian wetlands. Information is most available on impacts of hydrologic alteration, acidification, and nutrients, and least on impacts of salinization, sedimentation, thermal alteration, and habitat fragmentation.

Even qualitative lists of "expected" aquatic invertebrates in wetlands of various types do not appear to have been compiled, either nationally or by individual states. The USFWS has begun to compile such lists (pers. comm., Buck Reed, USFWS, St. Petersburg, FL) and some publications in the "community profile" series of the USFWS (Appendix C) mention particular taxa known to occur in wetlands.

Table 12. Examples of Invertebrate Density and Biomass Estimates from Wetlands.

BIOMASS (g/m^2)

State	type*	N	min.value	max.value	citation
AR**	L			15.0	Lowery et al. 1987
AR**	Pfo			mean=8.50	Cobb et al. 1984
IA	L			1.3	Tebo 1955
LA	Pab	48	0.5	15.7	Sklar 1985
LA	Pab	?	4.2	6.8	Sklar & Conner 1983
MS	Pfo	?		3.2	Baker et al. 1988
WA	Pfo	18	2.5	5.7	Meehan-Martin & Swanson 1983
WI	Pem	?	0.6	1706	Schmal & Sanders 1978
SD	Pem	220	1.3	8.5	Broschart & Linder 1986
CA	P(fen)	0.9		8.5	Erman & Erman 1975

DENSITY (number/m^2)

State	type	N	min.value	max.value	citation
AR*	L			>10,000	Lowry et al. 1987
AR***	Pfo			mean=2967	Cobb et al. 1984
CA	Pem	230	6952	23,857	Erman and Erman 1975
FL	Pfo			1,102	Brightman 1984
FL	Pfo			2.5	Brightman 1984
IA	Pem			>20,000	Voights 1976
IA	L			4,108	Tebo 1955
KS	Pem	?	508	18,676	Kansas Biol. Surv. 1987
LA	Pab	?		76,990	Sklar and Conner 1979
LA	Pfo			16,198	Sklar 1985
LA	Pfo			12.5	Sklar 1985
LA	Pfo	?	-	16,000	Sklar & Conner 1979
LA	Pfo	70		mean=95/grab	Beck 1977a
	Pem	13		mean=2900/grab	Beck 1977a
MI	Lab			>10,000	Lowery et al. 1987
MO	Pfo			>9,000	Batema et al.
MS	Pfo	?	1675	9248	Baker et al. 1988
GA	Pab	?		mean=12,093/m^2	Smock & Stoneburner 1980
MO	Pfo	4		5045	Batema et al. 1985
IL-MO	L	33	247	4321	Jones et al. 1985
KY	Pem	84	739	5143	Bosserman & Hill 1985
NJ	Pem	?	196	335,547	Gatter 1986
WI	Pem	?		35,730	Schmal et al. 1978
MI	Lab	?	7665	13,243	Hiltunen & Manny 1982
SD	Pem	175	584	5929	Benson & Hudson 1975
SD	Pem	220	3533	15,193	Broschart and Linder 1986
OR	R	64	33	15,700	Kreis & Johnson 1968
OR	Pem	?	11	1745	Fishman 1989

* wetland codes (Cowardin et al. 1979): Pab=palustrine aquatic bed, Pem=palustrine emergent, Pfo=palustrine forested, L=lacustrine, R=riverine

** includes TN and MS

***includes LA, MS, and MO

9.0 WETLAND FISH COMMUNITIES

This discussion includes both adult and larval fish, both game and nongame species. Few freshwater fish spend their entire life in wetlands, and wetlands that seldom contain surface water (e.g., raised bogs) do not usually have fish. Although fish community structure has been widely described in lakes and rivers, and "indices of ecological integrity" which integrate community data have been developed and tested (Karr 1981), such efforts have not yet been transferred to wetlands. Advantages and disadvantages of use of fish as indicators are shown in Appendix A. The paper by Munkittrick and Dixon (1989) provides further discussion of the value of fish as indicators of ecosystem condition. They assert that fish populations, in general, respond to reduced food resources initially by a decline in fecundity, followed by reduced condition factor, an increase in mean age, and finally a drop in population level. They suggest that these characteristics might be used to indicate the "health" of a particular population, and in some cases, the types of stress that are impairing population health.

9.1 USE AS INDICATORS

Enrichment/Eutrophication. Nutrient enrichment can result in increased fish biomass (Colby et al. 1972, Gascon and Leggett 1977) and altered species diversity (Nakashima et al. 1977) in lakeshore wetlands. Increased biomass may result from increased biomass of invertebrate fish foods, these having increased as a result of increased attachment surfaces and detritus provided by nutrient-induced expansion of submersed wetland plant beds (Pardue 1973). If fish food is already abundant, eutrophication may result in population increases in addition to biomass increases (Nakashima and Leggett 1975). Omnivorous species may benefit the most from the increase in submersed plants (Camp, Dresser and McKee 1989). Walleye (Stizostedion vitreum) and Mosquitofish (Gambusia) are two of dozens of wetland species that tolerate eutrophic conditions (Dawson and Hellenthal 1986), but few species occur exclusively in eutrophic waters.

Organic Loading/Reduced DO. Among northern lacustrine wetlands, Rahel (1984) reported that the ratio of cyprinid to centrarchid fish was greater where winter anoxia occurred. Rivers downstream from sewage and industrial waste outfalls showed a decline in fish community richness in Illinois (Lewis et al. 1981) and in Louisiana (Gunning and Suttkus 1984). In the latter study, two species of darter, Ammocrypta vivax and Etheostoma histrio, were particularly intolerant of the effluents. A southern wetland exposed to treated wastewater experienced increased fish productivity and decreased fish species richness (Camp, Dresser, and McKee 1989). Fish habitat in another wetland, a cypress pond in Florida, was degraded by wastewater effluent (Jetter and Harris 1976).

The State of Florida's regulations for discharge of treated wastewater into wooded wetlands specify that the biomass of sport-commercial or forage fish shall not be allowed to decline by more than 10%; exceptions may be allowed if such declines can be attributed, through analysis of covariance, to other factors. The State also specifies that the biomass of rough fish shall not increase more than 25% unless the ratio of sport and commercial fish to rough fish is maintained; sampling protocols are specified. Florida regulations consider the following fish taxa to be most tolerant of treated wastewater: suckers (all Catostomidae), tilapia (all Chichlidae), gar (Lepisosteidae), bowfin (Amia calva), grass carp (Ctenopharyngodon idella), common carp (Cyprinus carpio), and gizzard shad (Dorosoma cepedianum). Table 13 includes some other species that tolerate relatively low levels of dissolved oxygen.

Table 13. Examples of Wetland Fish Species That Tolerate Low Dissolved Oxygen.

Compiled from the ERAPT database (Dawson and Hellenthal 1986). Note that these species may occur as well in wetlands that are NOT anoxic, although usually in smaller proportion relative to other species.

Amia calva	Bowfin
Cyprinus carpio	Common Carp
Eriomyzon sucetta	Lake Chubsucker
Etheostoma nigrum	Johnny Darter
Ictalurus melas	Black Bullhead
Ictalurus natalis	Yellow Bullhead
Ictalurus nebulosus	Brown Bullhead
Moxostoma carinatum	River Redhorse
Notemigonus crysoleucas	Golden Shiner
Notropis buchanani	Ghost Shiner
Notropis heterodon	Blackjaw Shiner
Notropis heterolepis	Blacknose Shiner
Noturus gyrnus	Tadpole Madtom
Umbra limi	Central Mudminnow

Contaminant Toxicity. Declines in species richness and density of fish as a result of contaminants (oil, heavy metals, pesticides, etc.) have been widely documented in lakes and streams, but less often in wetlands. There exists a wealth of toxicological data from laboratory bioassays and tissue analyses. These include Johnson and Finley (1980), USEPA (1986), USEPA's "AQUIRE" database, and the US Fish and Wildlife Service's "Contaminant Hazard Reviews" series that summarizes data on arsenic, cadmium, chromium, lead, mercury, selenium, mirex, carbofuran, toxaphene, PCBs, and chlorpyrifos. However, relatively few field data are available for judging which wetland species are most sensitive.

Acidification. Acidity clearly affects fish species richness in lacustrine wetlands (Jackson and Harvey 1989, Rahel and Magnuson 1983, Tonn and Magnuson 1982). Fish species richness declined among a series of lacustrine wetlands with progressively more acidic conditions (Rahel 1984, 1986). Various reviews (e.g., Ford 1989, Hastings 1984, Wiener et al. 1983) indicate that, in northern lakes and streams, species most susceptible to the effects of acidification include lake trout, brook trout, Atlantic salmon, smallmouth bass, walleye, burbot, and common shiner and various other species of minnows. Data on acidification effects in other regions and wetland types are limited.

Salinization. No quantitative, published information was found concerning the effects of salinization of wetlands on community structure of indigenous fishes.

Sedimentation/Burial. No quantitative information was found concerning the effects of sedimentation in wetlands on community structure of indigenous fishes. It is widely documented that one common wetland fish--carp, Cyprinus carpio--resuspends deposited sediments and in doing so, may alter community structure of wetland plants and invertebrates, as well as fish. Since the feeding and reproductive habits of most fish are well-documented, it might be possible to detect gross sedimentation by the density-weighted ratio of sediment-feeding/breeding species to intolerant species.

Vegetation removal. Removal of canopy of forested wetlands generally results in increased algal production and possible increases in herbaceous wetland plants. Removal of submersed macrophytes (e.g., "aquatic weed control") may similarly increase algae. As vegetation is thinned, herbivorous fish species can increase and those that depend on macrophytes for cover can decrease disproportionately (e.g., Homer and Williams 1986, Wiley et al. 1984). However, total abundance and biomass may change little (e.g., Boschung and O'Neil, Mikol 1985, Wile 1978), and with the removal of vegetation the juveniles of some species may become more vulnerable to predation (Peterson 1982).

In contrast, if submersed vegetation becomes too dense, species richness can decline. For example, Lyons (1989) presents data supporting the theory that extirpation of many shiner, darter, and minnow species was caused by invasion and excessive growth (as a consequence of of the exotic milfoil, Myriophyllum spicatum, in Lake Mendota, Wisconsin. Some experimental studies of macrophyte removal have shown declines in total forage fish standing crop, but increases in growth rates, at least initially, of predatory (piscivore) fish; density of six of eight sunfish species declined while density of two cyprinids increased (Bettoli 1987).

Physical alteration of channel structure within wetlands can reduce fish biomass and total production, both within riverine wetlands (e.g., Arner et al. 1976, Portt et al. 1986) and within lacustrine wetlands (e.g., Eadie and Keast 1984). Species assemblages also shift. Poe et al. (1986) suggested that percid-cyprinid-cyprinodontid assemblages had a stronger need for diverse habitats and a lower tolerance for habitat alteration than did assemblages of centrarchids. These investigators found that the percid-cyprinid-cyprinodontid assemblage dominated an area with an undisturbed littoral zone, high water quality and high species richness of aquatic macrophytes. A nearby altered site with bulkheaded shoreline, dredged area, degraded water quality, and low species richness of aquatic macrophytes was dominated by a centrarchid assemblage. Moring et al. (1985) found brook trout to be particularly sensitive to canopy removal in western floodplain wetlands. Brook trout were replaced by a greater dominance of white sucker, northern redbelly dace, blacknose dace, creek chub, and common shiner.

Abundance of fish larvae in a southeastern floodplain swamp stream was found to be 16 times higher in macrophyte beds than in open channels during the daylight hours (Paller 1987). Durocher et al. (1984) found a highly significant positive relationship ($P<0.01$) between percent submerged vegetation and largemouth bass (Micropterus salmoides). Any reduction below 20 percent of the total lake coverage of vegetation caused a decrease in recruitment and standing stock of bass.

Turbidity/Shade. Increased turbidity, especially when it occurs over extended periods, generally decreases fish species richness and alters species composition (Menzel et al. 1984). Slight or moderate, seasonal increases in turbidity may or may not change fish density and biomass. Species commonly associated with elevated turbidity include carp, carpsuckers, black bullhead, green sunfish, and others (Menzel et al. 1984). Species apparently intolerant of elevated turbidity include fantail darter, smallmouth bass, northern hogsucker, rosyface shiner, hornyhead chub, southern redbelly dace, black redhorse, brook stickleback (Menzel et al. 1984) and many others listed in Plafkin et al. (1989). Also see above discussions of sedimentation/turbidity and vegetation removal.

Thermal Alteration. No quantitative information was found concerning the effects of thermal alteration on community structure of fishes specifically in wetlands. A shift toward warmer-water assemblages, e.g., carp, downstream from heated discharges seems inevitable.

Inundation/Dehydration. Virtually all fish depend on shallow-water habitats (i.e., generally wetlands) at some point in their life history. Some species depend more strongly than others on shallow areas and floodplain wetlands for feeding and reproduction. The proportion of highly-dependent species could theoretically be used as one indicator of hydrologic alteration of a wetland system.

Inundation alters the spatial and temporal distribution of suitable habitat, with unpredictable effects on floodplain-dependent species. Effects depend in part on habitat structure and soil chemistry of the areas being flooded, and whether inundation increases the exposure of isolated populations to predators or aggressive competitors. In southeastern floodplain wetlands many fish species benefit if water levels remain stable during the spawning period following seasonal inundation (e.g., Liston and Chubb 1984, Miranda et al. 1984). In the Florida Everglades, stable water levels resulted in increased fish community richness, diversity, biomass, average size of fish, and proportion of carnivorous species; however, fish density decreased (Kushlan 1976). In Mississippi River floodplain ponds, "days flooded" was the most significant factor in a multivariate regression for explaining total community biomass and biomass of catastomids, clupeids, crappies, cyprinids, and ictalurids; flooding in the sampled wetlands ranged from 24 to 115 days annually, with a mean of 81 (Cobb et al. 1984).

Dehydration reduces wetland fish diversity if it results in (for example) stranding of fish, anoxic conditions, cutting off of access, increased vulnerability to terrestrial predators, reduced area of productive periodically flooded areas, or altered food supply. However, periods of higher precipitation that follow droughts (or periods of inundation following partial drawdown) can result in increased fish production in wetlands; this could be due to increased nutrient availability or temporary elimination by drought of large competing or predatory invertebrates such as dragonfly larvae (Freeman 1989).

Where hydrologic alterations occur, the seasonality of their effects is critical in determining the effect they will have on fish community structure. Species considered by Mundy and Boschung (1981) to be most likely to decline with impoundment in Alabama floodplain wetlands were as follows: Bluehead Chub, Striped Shiner, Creek Chub, Creek Chubsucker, Frecklebelly Madtom, Crystal Darter, Scaly Sand Darter, and Redfin Darter.

Species that are most dependent on wetland portions of larger water bodies might be identified from existing regional literature (e.g., Crance 1988, Giese et al. 1987, Kwak 1988, Liston and Chubb 1984, Ross and Baker 1983, Tarplee 1975, Walker et al. 1985b) as well as from results of several ongoing studies of floodplain fish communities, e.g., studies being conducted by the Cooperative Fisheries Research Unit at Auburn University; the Corps of Engineers Waterways Experiment Station in Vicksburg, Mississippi; the U.S. Geological Survey in Tallahassee, Florida, and others.

Wetlands that normally contain surface waters but then are briefly dehydrated can, upon reflooding, support exceptionally high productivity and biomass of fish (Wegener et al. 1974, Welcomme 1979). However, this assumes fish have access into and out of the wetland as water levels change, and that sediments do not contain significant levels of oxidizable contaminants. Severely fluctuating water levels (i.e., causing repeated exposure of sediments every few hours or days) associated with hydropower generation or boat wakes can kill fish larvae (Holland 1987).

Fragmentation of Habitat. We found no explicit information on wetland fish community response to fragmentation of regional wetland resources. One can surmise that as the distance between wetlands containing fish becomes greater, and/or hydrologic connections become severed by dehydration or dams, species most dependent on floodplain habitats and/or which do not disperse easily might be most affected. The magnitude of the effect may depend on the size and intrinsic habitat heterogeneity of the wetlands that are being fragmented.

Availability of patches of relatively unaltered habitat with natural flow regimes, such as may occur in lower-order tributaries, can help sustain mainstem fish populations even when mainstem habitats are periodically subjected to pollution or extreme hydrologic alteration (e.g., Gammon and Reidy 1981). The distances between such patches may be important. In streams, individual non-anadromous fish over the course of a year seldom disperse more than a kilometer (Hill and Grossman 1987); however, substantially greater mobility (frequent movements of up to 12.7 km) was reported for fish inhabiting North Carolina floodplain wetlands (Whitehurst 1981).

In lakes, fish species diversity increases with increasing surface area and length of shoreline (Barbour and Brown 1974, Moyle and Cech 1982, Tonn and Magnuson 1982), probably as a result of increased habitat heterogeneity and thermal stratification (Eadie and Keast 1984).

Other Human Presence. Sport and commercial fishing comprise an obvious impact to certain wetland fish species, in some cases at the population level.

9.2 SAMPLING METHODS AND EQUIPMENT

Some factors that could be important to measure and (if possible) standardize among wetlands when monitoring anthropogenic effects on community structure of fishes include:

> hydrologic access, water depth, winter ice cover, conductivity and baseline chemistry of waters and sediments (especially pH and dissolved oxygen), sediment type, current velocity, fishing pressure (harvest), stream order or ratio of discharge to watershed size (riverine wetlands), shade, amount and distribution of cover (logs, undercut banks, etc.), ratio of open water to vegetated wetland, and the duration, frequency, and seasonal timing of regular inundation, as well as time elapsed since the last severe inundation or drought.

Methods for sampling fish communities are described in Kushlan 1974b, Nielsen and Johnson 1985, Plafkin et al. 1989, Welcomme 1979, and many others.

Often, fish can be found in wetlands only during certain seasons of the year. If wetlands can be sampled only once, then the period just after seasonal rise in water levels, if it coincides with favorable temperatures, is usually recommended. In most regions, numbers of easily identifiable fish will be greatest late in the season due to annual recruitment of juveniles. However, caution is needed to time sampling to coincide with phenologies of particular taxa. Significant, regular events of fish life histories include migration, dispersal, territory establishment, spawning, and development (Brooks 1989).

Larval fish sampling is best accomplished at night to minimize sample bias due to fish avoiding the sampling gear (Chubb and Liston 1986). Schramm and Pennington (1981) also suggested nighttime sampling and showed a maximum of larvae at dusk, high diversity at night and dawn, and low diversity in the daytime. Nighttime samples were particularly important for collection of hiodontids, ictalurids, and percichthyids.

Equipment used in wetlands for fish sampling potentially includes seines, nets, trawls, electrofishing, ichthyocides, and various types of pot gear (Hocutt 1978, Nielsen and Johnson 1985, Plafkin et al. 1989). For sampling larval and egg stages, push-nets and modified plankton nets are often used (e.g., Meador and Bulak 1987), while in dense vegetation, suction pumps and light traps are often used. A study by Pardue and Huish (1981) evaluated techniques for collecting adult fish in forested wetland streams, and found that no single technique collected all species. Thus, they are best used in combination. Scientific collecting permits, available from state fish agencies, are generally required.

Electrofishing. temporarily stuns fish and thus allows them to be scooped into a bucket, identified and measured, and quickly released. Electrofishing equipment is commercially available, and permits for scientific collecting are typically required from state agencies. If the sampled wetland has clearly defined inlets and outlets, these may be blocked with nets to prevent fish from escaping ahead of the electrical field. Repeated passes are typically made. Electrical currents are not always used to stun fish; they may also be used to guide fish into nets or block their escape from a seining area (Nielsen and Johnson 1985).

Electrofishing can quickly obtain fish from many wetland habitats that are difficult to sample with nets, e.g., undercut banks, submersed plant beds. For quantification, data are best expressed as number of fish per unit area shocked. However, quantitative accuracy is good only for narrow, non-turbid channels. Morgan et al. (1988) reported that the effectiveness of electrofishing generally decreased as plant density or turbidity increased, due to the difficulty in locating and retrieving stunned fishes. Backpack shockers may be too bulky and unsafe for use in wetlands with extensive debris, very soft substrate, and/or ice. Boat-mounted shockers are limited by shallow water depths, debris, and ice.

It is often difficult in wetlands to confine the area being sampled, so fish may flee the advancing electrical field. Also, fish stunned by shockers are not necessarily representative of the general fish community. Collections tend to be biased toward larger individuals and species. Larvae are not captured. Some studies suggest that catchability of fish declines with successive passes through a wetland, with the effects lasting up to 24 hours.

Sampling efficiency can also be influenced by water quality. Pulsating, direct current (DC) units are effective in perhaps the widest range of conditions, but in the "soft" waters of many wetlands (particularly bog streams), AC units with outputs exceeding 500 volts might work just as well. Some investigators in small, confined soft-water wetlands have increased shocker effectiveness by placing salt blocks in the water, which increases conductivity. Extremely high conductivity can reduce effectiveness as well. Bosserman and Hill (1985) found that shockers were not effective in waters made highly conductive by acid mine drainage.

Seines. are robust nets, several meters long and with a width usually equal or greater than water depth, that are pulled by people or boats through shallow areas to confine and capture fish (often by herding them toward shore). When aquatic plants and debris interfere, seines can instead be placed in adjoining open areas and fish herded into the seines for capture (Nielsen and Johnson 1985). Seines are too ineffective for accurately estimating fish densities, but may allow a fair estimation of species richness and of relative dominance of species. Leidy and Fiedler (1985) used a 3 m long seine of 6 mm mesh to sample shallow streams. Ohio streams were sampled using a 4 ft x 8 ft "Common Sense" minnow sein with 1/8 inch mesh; about 30 seine hauls were required for thorough sampling (Tramer and Rogers 1973). For wetlands, Hocutt (1978) recommended the 5 ft x 10 ft "Common Sense" seine with 1/8 inch mesh. A mesh size of 1/8 inch mesh was recommended to capture smaller species and/or life stages. In situations where larger fish may outswim smaller seines, monofilament gill nets can be used for seining.

Sweep nets. (dip nets) can be used to capture fish as well as invertebrates. They can be effective for qualitative sampling in very confined, shallow, clearwater pools. Walker et al. (1985b) had limited success when dip-netting floodplain fish immobilized with spotlights at night. Studies using sweep nets include those by Leidy and Fiedler (1985) and Chubb and Liston (1986).

Other types of nets. are used to catch wetland fish, in a passive manner. All nets tend to be selective due to their design and thus usually provide the best catch results when used in combination. **Fyke nets** have been used to sample fish in wetlands (Nielsen and Johnson 1985, Swales 1982, Tonn 1985). Wetland vegetation is sometimes removed in a small area to make room for the net. **Gill nets** can be used to take a variety of fish and can be adapted to different depths (Hocutt 1978). These nets are very effective in wetlands, and are highly selective for particular size classes and species (Pennington et al. 1981). Gill nets of five mesh sizes between 2.54 and 12.7 cm were placed near shore in Ohio riverine areas by Hassel et al. (1988) and checked after 24 hours. Gill net selectivity produced catches dominated by relatively large species such as longnose gar and channel catfish. **Trammel nets** purportedly are less size selective than gill nets (Pennington et al. 1981), but select for fish species with rough surfaces and protrusions (Nielsen and Johnson 1985). Although traditional **trawl nets** are not effectively used in wetlands, Herke (1969) described a boat-mounted push-trawl useful for sampling marshes.

Lift nets. constructed with rectangular frames, hoops, or spreaders are set on the bottom below the water surface, then lifted to capture small schooling fish (Nielsen and Johnson 1985). Camp, Dresser and McKee (1989) reported on a lift net specifically designed for use in forested wetland systems. The 1 meter square net was made of two weighted PVC loops (a top and bottom) with netting of black fiberglass screen. When fully extended, the bottomless net measured 39.4 inches x 39.4 inches x 36 inches with a 6 inch flap along the base. The bottom frame was attached to the substrate and the top frame was connected to a rope and pulley system to allow the trap to be sprung (lifted) from a remote location without frightening any fish within the net. Small portable drop nets were used by Freeman et al. (1984) to sample fish in a heavily vegetated freshwater wetland. These collected significantly more fish per unit area than did seining. Large drop nets suffer problems of mobility, and when designed to be portable, create disturbance by movement and shadows (Freeman et al. 1984).

Pot gear. (fish traps) of wood, wire mesh, and/or acrylic plastic have been routinely used by several experimenters. Traps can be used in a variety of areas of moderate depth and/or heavy cover, and when baited, are strongly selective for particular species and size classes (Pennington et al. 1981). Studies that used fish traps in wetlands include Finger and Stewart (1988), Tonn and Magnuson (1982), Walker et al. 1985b.

Ichthyocides are poisons (preferably biodegradable) that can be used to destructively sample the entire fish population of a wetland. They are undoubtably the most efficient tools for obtaining both quantitative and qualitative fish samples. However, when used by inexperienced collectors, problems may outweigh benefits (Hocutt 1978). Examples of use in wetlands include studies by Durocher et al. (1984) and Walker et al. (1985b).

Also, radiotelemetric methods can be used to track individuals (e.g., Savitz et al. 1983) and estimate potential wetland dependency.

9.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

In general, quantitative data on wetland fish community structure has not been uniformly collected from a series of statistically representative wetlands in any region of the country. Thus, it is currently impossible to state what are "normal" levels for parameters such as fish density, species richness, biomass, Index of Biotic Integrity (IBI, Karr 1981) and their temporal and spatial variability, in any type of wetland.

A data set that is perhaps the closest to meeting this objective was collected from a series of relatively pristine Arkansas rivers that are mostly bordered by wetlands (Giese et al. 1987). These fish data were collected in part from streams flowing through relatively pristine floodplain wetlands, and thus help serve as a regional baseline for bottomland hardwood wetlands. Although data on fish density were not developed, up to 36 species per stream were found and community structure of relatively pristine streams was defined. In nearby Kentucky, a riverine slough wetland supported at least 12 species (Bosserman and Hill 1985).

In submersed wetland plant beds, up to 255 fish per $10m^2$ may be present (Morgan et al. 1988). On the floodplain of the Kankakee River in Illinois, 481 fish were captured during 4800 hours of trapping (Kwak 1988). In one of the few studies of larval fish communities, Chubb and Liston (1986) reported densities of up to 32.2 larvae per m3 from Great Lakes emergent wetlands.

For stream fish studies, coefficients of spatial variation have ranged from about 50 to 150 percent (Eberhardt 1978). In submersed vegetation, this coefficient may range from 9 to 80 percent (Morgan et al. 1988). Studies that have compared fish communities among wetlands (spatial variation) have largely been conducted along the lower Mississippi River, and include:

Baker et al. 1988, Cobb and Clark 1981, Cobb et al. 1984, Conner et al. 1983, Felley and Hill 1983, Hall 1979, Lowery et al. 1987, Mathis et al. 1981, Pennington et al. 1980, and others.

Only a few studies (Clady 1976, Freeman 1989, Kushlan 1976, Lyons 1989) have quantified year--to-year or long-term variation in fish community structure in wetlands, but conceivably unpublished data may be available from sites of the U.S. Department of Energy's National Environmental Research Park system, as well as the following sites of the National Science Foundation's Long Term Ecological Research (LTER) program (that contain studied wetlands): Illinois Pool 19 site, Illinois-Mississippi Rivers sites, New Hampshire Hubbard Brook riparian forest, Oregon Andrews Experimental Forest riparian forest, and Michigan Kellogg Biological Station site. Temporal (year-to-year) variation in western riparian fish communities was quantified by Platts and Nelson (1988). Although state fishery agencies undoubtedly have long-term data on average biomass or length of captured game fish, these data may not have been systematically collected from wetland sites, and do not include all wetland fish species.

Quantitative data on community composition of wetland fish appears to be most available for lacustrine aquatic bed (herbaceous) wetlands, western riparian wetlands, and southeastern bottomland hardwood systems. Apparently such data are least available for riverine herbaceous wetlands and for riparian wetlands in other regions.

Even qualitative lists of "expected" fish in wetlands of various types do not appear to have been compiled, although regional distribution of fish is relatively well-documented (e.g., Hocutt and Wiley 1988; Lee et al. 1980). Some publications in the "community profile" series of the USFWS (Appendix C) mention particular taxa known to occur in wetlands, and wetland fish are listed in the ERAPT database (Dawson and Hellenthal 1986), in Niering (1985), and in the "Vertebrate Characterization Abstracts" database managed by The Nature Conservancy and various state Natural Heritage Programs. Quantitative data are generally most available for harvested species, and less available for non-game species.

10.0 WETLAND AMPHIBIANS AND REPTILES

10.1 USE AS INDICATORS

This discussion addresses the monitoring of "herptiles"--turtles, frogs, salamanders, snakes, crocodilians, and lizards that occur in wetlands. The life histories and requirements of amphibians differ greatly from those of reptiles, and species within each group also differ significantly. Most amphibian species and many reptiles spend all or critical parts of their life in wetlands. However, with only a few exceptions (Brooks and Croonquist 1990, Corn and Bury 1989), their responses to anthropogenic stressors in wetlands have barely been studied in the United States at the community level. Most recent ecological research on herptiles can be characterized as assessments of the occurrence and abundance of particular species in specific micro-habitats. Advantages and disadvantages of use of herptiles as indicators are shown in Appendix A. A possible approach for using assemblages of anuran amphibian species (frogs and toads) as indicators of wetland condition is described by Beiswenger (1988).

Enrichment/Eutrophication. The effects of enrichment on overall community structure of herptiles apparently have not been documented in wetlands, and indicator assemblages of "most sensitive species" remain speculative for this stressor. In southern England, Beebee (1987) found that the bullfrog, Bufo calamita, consistently selected the more eutrophic wetlands.

Organic Loading/Reduced DO. The effects of severe organic loading, e.g., from wastewater outfalls, on overall community structure of herptiles apparently have not been documented in wetlands, and indicator assemblages of "most sensitive species" remain undefined for this stressor. Toxicological data were reviewed by Birge et al. (1980). Anderson (1965) noted that a moderate amount of sanitary sewage pollution seemingly increased the dominance of soft-shelled and snapping turtles in parts of the Missouri and Mississippi Rivers, but heavy industrial waste nearly eradicated turtles for miles downstream, especially the Ouachita map turtle, in part a mollusk-eater.

Contaminant Toxicity. The effects of heavy metals, pesticides, oil, and other contaminants on the overall community structure of herptiles apparently have seldom been documented in wetlands, and indicator assemblages of "most sensitive species" remain speculative for such stressors. Speculation about causes of regionwide or even global declines in several wetland amphibians (e.g., northern leopard frog, boreal toad, spotted frog, tiger salamander in the Rocky Mountains) has often focused on either (a) effects of airborne contaminants on growth and development of tadpoles (Phillips 1990), or (b) effects of increased ultraviolet-B radiation as a result of trophospheric ozone depletion, since such declines have been noted in otherwise seemingly pristine wetlands.

Some laboratory based toxicological data for individual species may be found in USEPA (1986), EPA's "AQUIRE" database, and the U.S. Fish and Wildlife Service's "Contaminant Hazard Reviews" series that summarizes data on arsenic, cadmium, chromium, lead, mercury, selenium, mirex, carbofuran, toxaphene, PCBs, and chlorpyrifos. However, relatively few field data are available for judging which wetland species are most sensitive.

Acidification. Larval stages of amphibians have been suspected of being highly sensitive to acidification effects. Although impacts at the species level have most often been reported (e.g., Clark 1986a,b, Corn and Fogelman 1984), acidification impacts on the overall community structure of herptiles have been documented in wetlands only recently (e.g., Corn et al. 1989, Leuven et al. 1986). Turner and Fowler (1981) found significantly fewer species in wetlands with pH of less than 5.5.

A few species, e.g., wood frog (Rana sylvatica), are known to be particularly tolerant of acidic pH in bogs (Karns 1984). However, most amphibians require a pH of higher than 4.5 to 5.0 for embryo survival and metamorphosis (Corn et al. 1989, Freda 1986, Freda and Dunson 1986, Gosner and Black 1957, Karns 1984, Pierce 1985).

Acidic conditions in surface-mine (constructed) wetlands were implicated as a reason for reduced amphibian use by Hepp (1987). Based on a single pH measurement from each surface-mine pond, the mean pH at which various species occurred was given by Turner and Fowler (1981) as follows:

	pH	# of ponds
Northern Spring Peeper	5.2	16
Pickerel Frog	5.42	11
Red-spotted Newt	5.80	8
Gray Tree Frog	5.96	9
Bullfrog	5.91	6
American/Fowler's Toad	5.97	7
Northern Cricket Frog	6.00	1
Wood Frog	6.25	2
Green Frog	6.26	8
Spotted Salamander	6.32	8
Upland Chorus Frog	6.33	8

Similar types of data are presented by Clark (1986b) for Ontario wetlands.

Most of the true frogs are thought to be especially sensitive to acidic precipitation because they respire through their skin. During foggy periods such respiration may occur while they are out of the water. At such times, they may be directly exposed to airborne contaminants.

Salinization. The effects of salinization, e.g., from irrigation return water and oil drilling wastes, on overall community structure of herptiles apparently have not been documented in wetlands, and indicator assemblages of "most sensitive species" remain undefined for monitoring salinization. In softwater lakes and streams, moderate increases in water hardness and alkalinity can result in increased amphibian densities (Hepp 1987).

Sedimentation/Burial. Moderate increases in soft bottom sediments can increase habitat for overwintering turtles. However, excessive sedimentation can smother eggs of many amphibians and alter food sources. The North American dusky salamander (Desmognathus fuscus) and the spring salamander (Gyrinophilus porphyriticus) are reportedly very sensitive to effects of bank erosion, sedimentation, and turbidity (Campbell 1974, Orser and Shure 1972). However, the effects of sedimentation/ burial (e.g., of amphibian eggs) on overall community structure of herptiles apparently have not been documented in wetlands, and indicator assemblages of "most sensitive species" remain speculative for this stressor.

Turbidity/Shade, Vegetation Removal. Many herptiles are sensitive to the presence and type of vegetation and its juxtapositioning with open water, particularly in arid regions. In Colorado River riparian zones, lizards were most abundant in shoreline habitats, moderately dense in riparian habitats, and least dense in non-riparian or upland habitats; densities depended on insects inhabiting herbaceous debris heaps and litter piles washed up by the river (Jones and Glinski 1985, Warren and Schwalbe 1985). In more humid Oregon watersheds, amphibian richness, density, and biomass were less in logged watersheds than in unlogged watersheds, particularly when vegetation removal occurred primarily in headwater areas (Corn and Bury 1989). Herptile richness was also less in Pennsyvania watersheds with disturbed stream corridors than in those with intact riparian vegetation (Croonquist 1990). Use of riverine wetlands by herpetofaunas has been positively related to number of cover types, sinuosity, circumneutral pH, and gradual shoreline slopes (Hill 1986). Richness of breeding frogs may be related also to the variety of herbaceous plant forms in a wetland (e.g., Diaz-Paniagua 1987).

The community composition of Minnesota amphibians was found to be correlated with wetland vegetation form. The leopard frog (Rana pipiens) was found most frequently in sedge mat and less commonly in the very wet tamarack zone. The wood frog (Rana sylvatica) was found primarily in the fir-ash zone with lesser numbers in the spruce and tamarack. Spring peeper (Hyla crucifer) and swamp frog (Pseudacris nigrita) were found in the two zones most distant from the pond, spruce and fir-ash (Marshall and Buell 1955).

Despite these initial efforts, indicator assemblages of "most sensitive species" of herptiles remain speculative in most of the U.S. for monitoring effects of vegetation removal, and the effects of vegetation removal on overall community structure of herptiles apparently have not been documented in wetlands.

Thermal alteration. Herptiles, as ectotherms, are particularly sensitive to thermal alteration of wetlands. Although a vast literature exists describing thermal preferenda of individual species, the effects of thermal alteration on overall community structure of herptiles apparently have seldom been documented in wetlands. Lack of comparative studies has resulted in a lack of information on most-sensitive indicator assemblages.

Dehydration/Inundation. Changes in wetland water level alter the quantity and quality of herptile habitat, and may trigger immigration, emmigration, and breeding of particular species and their predators (Pechmann et al. 1988). The effects of dehydration may be particularly severe if dehydration occurs during herptile hibernation, due to the effects of exposure and increased predation of eggs (Campbell 1974).

Impoundment has been reported to increase the regional populations of toads and turtles (Anderson 1965), or at least causes a shift in spatial distribution of habitat. However, inundation can reduce and alter the seasonal timing of flooding of downstream habitats. The resultant changes in vegetation and floodplain leaf litter accumulation can reduce both abundance and diversity of reptiles, as reported by Jones (1988) for Arizona riparian systems. Also, if inundation causes temporarily flooded wetlands to become connected to permanent waters, predatory fishes can gain access to the temporary wetlands, perhaps resulting in reductions in some amphibians (e.g., Dodd and Charest 1988). Temporary dehydration of wetlands may have the opposite effect, benefitting amphibians by reducing fish predation. The ratio of non-predatory to predatory salamanders can increase in wetlands following dry springtime conditions (Cortwright 1987).

Herptile taxa that characterize seasonally flooded wetlands or have terrestrial phases appear to resist effects of urbanization more than those that characterize permanently flooded wetlands or which spend their entire life cycle in wetlands (Minton 1968). In San Francisco, Banta and Morafka (1966) attributed the decline of the native California red-legged frog (Rana aurora draytoni) and the introduced leopard frog (Rana pipiens) to dehydration and filling of wetlands. Leopard frogs also declined in Colorado as a probable result of drying up of breeding ponds during a drought (Corn and Fogleman 1984). Vickers et al. (1985), studying aquatic and semi-aquatic amphibians in northern Florida cypress wetlands, found no change in mean numbers, numbers of species, or species diversity in ditched versus unditched wetlands. However, species richness declined and terrestrial species became more abundant with ditching.

Fragmentation of Habitat. We found no explicit documentation of herptile community response to fragmentation of regional wetland resources, although the presence of some individual species, e.g., spotted salamander, is known to sometimes depend on proximity to source ponds (Cortwright 1987). One can surmise that as the distance between wetlands containing herptiles becomes greater, and/or hydrologic connections become severed by dehydrated channels, dams, or (particularly) roads, species most dependent on wetlands and/or which do not disperse easily might be most affected (Campbell 1974, Croonquist 1990). In Oregon, Corn and Bury (1989) found that logging upstream from unlogged habitats had no effect on the presence, density, or biomass of any species inhabiting the unlogged habitat.

Other Human Presence. The introduction by humans of non-indigenous aggressive predators (e.g., bullfrog, snapping turtle, and several predatory fishes) into particular water systems has sometimes led to a decline in richness of indigenous frog communities (e.g., Hammerson 1982, Hayes and Jennings 1986, Moyle 1973).

10.2 SAMPLING METHODS AND EQUIPMENT

Some factors that could be important to measure and (if possible) standardize among wetlands when monitoring anthropogenic effects on community structure of herptiles include:

> water depth, temperature (site elevation, aspect), conductivity and baseline chemistry of waters and sediments (especially pH, DO, and suspended sediment), current velocity, stream order or ratio of discharge to watershed size (riverine wetlands), shade, amount and distribution of cover (logs, crevices, etc.), ratio of open water to vegetated wetland, extent of plant litter and rotting logs, vegetation type, and the duration, frequency, and seasonal timing of regular inundation, as well as time elapsed since the last severe inundation or drought.

Sampling methods for wetland herptile communities are described in Bury and Raphael 1988, Corn and Bury 1990, Halvorson 1984, Jones 1986, Scott 1982, Vogt and Hine 1982, and others.

Because amphibian distribution and abundance has strong ties to seasonal hydrologic phenomena and the capability of particular species for dispersal, the temporal and spatial variability in amphibian community structure strongly reflects these factors. As is the case with sampling macroinvertebrates whose communities are similarly dependent on ephemeral hydrologic events, sampling amphibian communities can require several repeated visits to a wetland to fully describe community composition. Nonetheless, Corn and Bury (1989) assert that, at least for riparian communities of the Pacific Northwest, amphibian population densities are usually stable in undisturbed habitat and serve as better indicators of habitat quality than do similar densities of birds or mammals.

Herptiles can be sampled during the mid- to late growing season when maximum numbers of developing juveniles are present. However, many species are easy to find only after the first few days of rain following a drought, during late-summer thunderstorms, during the first spring thaw in northern areas, during mid-day basking hours, or at night (Kaplan 1981). Occasionally, traditional winter hibernation areas can be located and used to count individuals representing a larger (but undefinable) area. For Arizona, Jones (1986) noted trapping was most effective in riparian habitats between May and July.

Methods used in wetlands for herptile sampling potentially include pitfall traps (often with drift fences and baited), visual belt transects, direct capture methods, and vocalization recording.

Pitfall traps and funnels are perhaps the most widely used mechanism for capturing herptiles (Jones 1986). Animals enter and cannot find the opening to escape. They are subsequently identified, counted, measured, and released. To reduce loss of trapped animals to predation, traps and funnels can be checked regularly (at least every other day) and can be shaded, and/or filled with sufficient moist plant litter to minimize physiologic stress to animals. Pitfall traps are impractical in many wetlands where the water table is so close to the land surface that pits fill rapidly with water.

Pitfall traps and funnels often produce more species per sampling effort than direct capture methods (Jones 1986). The size of the trap, baits used, and trap placement can affect the species that are caught. Trap and funnel methods can provide relatively quantitative data, when arranged systematically and level-of-effort (e.g., "trap-hours") is standardized.

They involve emplanting a container in the soil, either on the periphery of the wetland or within it (if surface water is absent), with the lip of the container placed flush with the ground surface. Herptiles stumble in and cannot climb the steep sides to escape. Because some species can drown if the container fills with rainwater, Jones (1986) recommends placing floatable material (e.g., styrofoam) in the container to reduce mortality.

Funnel openings are usually oriented toward land for greatest effectiveness. Hoop funnel traps are generally used for turtles, and other funnel traps are used (particularly in deeper wetlands) for catching salamanders, frogs, and occasionally snakes. A special kind of floating pitfall trap can be used to sample basking turtles (see Jones 1986 for description). Aquatic turtles in a Missouri marsh were captured using hoop and net traps. Traps were baited with sardines, local fish, tadpoles, frog, crayfish, dragonfly larvae, snails, and clams (Kofron and Schreiber 1987).

The efficiency of traps and funnels can be increased by channeling movements of herptiles in the direction of the trap or funnel. This is commonly done with "drift fences" (Gibbons and Bennett 1974). These are fences constructed of wire screen or polyethylene plastic, with lengths upwards of 15 m. Traps are placed at both ends of the drift fence, along the fence at various points, or at the junction of several intersecting fences. The bottom edge of the fence is emplanted in the ground, or at least no space is provided for herptiles to crawl under the fence.

Drift fences and pit traps can be more effective and less biased than log-turning, walking transects, electroshocking in streams, or searching and digging through litter. However, they are expensive; time and cost estimates for drift fence trapping are provided by Gibbons and Semlitsch (1982). Jones (1986) comments that, for quantifying herptile communities, drift fence/pitfall trap methods are less effective for frogs, toads, large snakes, terrestrial turtles, and salamanders than for small snakes and lizards.

Sizes and shapes of containers and associated drift fences and their configurations vary greatly, depending partly on target species and wetland type. Vickers et al. (1985) sampled in and around cypress ponds using arrays of four 7.6 m lengths of 0.75 m high, 6 mm polyethylene drift fences arranged perpendicularly and attached at the center. The fences were held upright with wooden laths and buried to 10 cm depth to avoid animals passing underneath. Two aluminum screen wire funnel traps, 75 cm long with 20 cm entrance funnels were placed beside each drift fence. To insure that a trap would always fall on the ecotone regardless of pond size, distance between arrays was standardized at one-half the pond radius. Working in peatland vegetation in Maine, Stockwell (1985) censused herptofauna in eight vegetation types -lagg, forested bog, wooded heath, shrub heath, moss lawns, pools, streamside meadow, and shrub thicket. Drift fences of free standing aluminum flashing were used as well as those of lath supported polyethylene. Pitfall traps were made of two #10 tin cans joined with tape and silicone. Funnels made from margarine tubs were used in the top of each trap to prevent escape and 2-3 cm of water placed in the bottom of each trap to prevent desiccation of captives. Similar traps were made by Jones and Glinski (1985) using double-deep 3 lb. coffee cans with a lid placed 15 cm over the top to prevent desiccation.

The above methods require multiple visits to a wetland, first to set up and later to check traps. Herptiles can also be monitored directly, that is, during a single visit, or without having to wait for traps to catch individuals. However, direct methods usually do not provide accurate quantitative data on abundance. Unless frequent visits are made and the correct microhabitats are searched at the proper times of year, direct methods are also unlikely to yield good estimates of species dominance or richness. However, they can provide a useful complement to trap methods, locating species that are not easily trapped.

The simplest type of direct search involves scanning a wetland with binoculars to observe the more obviously visible species such as basking turtles, frogs, and alligators. In some cases, floating egg masses of amphibians can also be detected visually and identified to species. Observational methods can be done formally, along defined transects. Searches on foot, perhaps employing many people shoulder-to-shoulder (e.g., Marshall and Buell 1955) have been used, but could be impractical and destructive of habitat in many wetlands. Long-handled nets can be used to surround logs and rocks as they are lifted to search for herptiles, so as to catch individual herptiles as they flee. In riverine wetlands, fine-mesh seines (see Fish section above) can be used for similar purposes.

To enhance opportunities for encountering herptiles during direct searches, electrofishing and identification of vocalizations and tracks can be used. Electrofishing methods (described in section 9, used in conjunction with sweep nets or seines, are particularly effective for retrieving larger salamanders and frogs. Because some species leave distinctive tracks, travel corridors can be searched periodically for tracks. Frogs can sometimes be located more easily at night, as their eyes reflect in the beam of a flashlight. Vocalizations of many frogs and toads are easily identified (commercially-available recordings are available to learn these) and can be used to augment observations. Frogs and toads can sometimes be induced to vocalize by introducing sharp, loud sounds or played-back tape recordings of vocalizations. Low-altitude overflights or aerial photography under favorable conditions can be used to identify alligator holes and paths.

10.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

In general, quantitative data on structure of the entire herptile community of wetlands has not been uniformly collected from a sufficiently large, statistically-drawn sample of wetlands in any region of the country. Thus, it is currently impossible to state what are "normal" levels for parameters such as herptile density, species richness, or biomass, and their temporal and spatial variability, in any type of wetland.

We found only a few published studies that quantified the entire herptile community (or a large proportion of it) across a region and/or among a set of wetlands:

> Brooks et al. 1985, 1987, 1989, Clark 1986b, Congdon et al. 1986, Corn and Bury 1989, Corn et al. 1989, Fowler et al. 1985, Gibbons and Semlitsch 1982, Jackle and Gatz 1985, Karns 1984, Hepp 1987, Jackle and Gatz 1985, Stockwell 1985, Turner and Fowler 1981, Ward 1988.

We found no journal articles that quantified year-to-year variation in the entire community structure of herptiles in wetlands, but conceivably such unpublished data may be available from sites of the U.S. Department of Energy's National Environmental Research Park system, and sites of the National Science Foundation's Long Term Ecological Research (LTER) program. Some studies (e.g., Corn et al. 1989) have featured qualitative re-checking of wetlands known in previous decades to have particular species, but probably could not be termed "long-term monitoring."

Quantitative data on community composition of wetland herptiles is virtually lacking from all regions except the Southeast, Southwestern riparian areas, and parts of the Northeast and Pacific Northwest. Information on impacts is limited mostly to studies of hydrologic effects and vegetation removal; especially little is known of impacts from contaminants, salinization, sedimentation, and habitat fragmentation.

Qualitative lists of "expected" herptiles have been compiled by statewide herptile atlas projects in Illinois, Kansas, Massachusetts, Maine, and perhaps other states, as well as by less comprehensive surveys in various smaller areas. Species that show highest affinity for wetlands of various types might be identified by consulting with local herpetologists, Niering (1985), and the "Vertebrate Characterization Abstracts" database managed by The Nature Conservancy and various state Natural Heritage Programs. Limited qualitative information may be available by wetland type from some of the "community profile" publication series of the USFWS (Appendix C).

11.0 WETLAND BIRD COMMUNITIES

11.1 USE AS INDICATORS

This discussion addresses the monitoring of wetland birds, e.g., waterfowl, shorebirds, wading birds, and wetland-dependent songbirds. The use of birds as environmental indicators is discussed by Morrison (1986), Reichholf (1976), and particularly, by Temple and Wiens (1989). Statistical aspects of regional bird trend analysis are discussed in Sauer and Droege (1990). Advantages and disadvantages of using birds as indicators are summarized in Appendix A.

Because most vertebrates use wetlands at some time during the year, defining what truly constitutes a "wetland-dependent" bird species is difficult. One could argue dependency based on diet, energetics/metabolism, requirement for a particular structural component found only in wetlands, or duration of seasonal use. As with some wetland plant groups, many degrees of dependency occur, from species that spend their entire life in wetlands to species that use wetlands opportunistically and/or for only brief periods. Species that may be casual users of wetlands of a particular type in one region may be obligate users of a different type in the same region, or of the same type in a different region. Dependency in highly altered landscapes may be less related to the intrinsic characteristics of wetlands than to the fact that little other undeveloped habitat remains, forcing species that normally occur less frequently in wetlands to use what remains, regardless of its condition. In such cases, bird density may be a poorer indicator of habitat quality (the ability of the habitat to sustain successfully reproducing individuals over the long-term) than measurements of population demographics or measurements made at the organism level (Van Horne 1983). An empirical approach for testing wetland-dependence of birds is demonstrated by Finch (1990).

Monitoring of wetland birds, particularly waterfowl, has been extensive in many regions. Wetland birds can be categorized as (a) those most strongly dependent on larval insects, non-insect aquatic invertebrates, amphibians, fish, and submersed plants, and (b) those most strongly dependent on adult (terrestrial) invertebrates, emergent plants, and rodents. In general, the former group--which includes waterfowl and wading birds--tends to respond more immediately to contamination and water level changes than does the latter group--which usually includes marsh wrens, certain warblers, red-winged blackbirds, and swallows. Diets (and thus, guild assignments) of particular species can be confirmed through stomach content analysis or, less destructively, through close-range, automated photography of nest visits. In general, though, habitat requirements, life histories, and species assemblages of wetland birds are relatively well-known. Still, information on community-level response to particular stressors has been difficult to collect, in part because most bird species--as very mobile organisms--may be better at integrating overall landscape conditions than they are at indicating the conditions in a particular wetland.

Enrichment/Eutrophication. The effects of enrichment on overall community structure of birds are poorly documented in wetlands, and indicator assemblages of "most sensitive species" remain mostly speculative for this stressor. Weller and Spatcher (1965) defined a species assemblage that inhabits a "late marsh" successional stage, and species that inhabit the upland transitional zones of wetlands are well-known. However, the dominance of these species assemblages may be related as much to physical factors (geomorphology, fire, extreme climate events) as to nutrient enrichment. For Great Lakes wetlands, Crowder and Bristow (1988) hypothesized the following series of events that might lead to a waterfowl decline as a result of eutrophication:

> "For the waterfowl, the effect of inshore eutrophication is thus an initial increase in food plants, a gradual replacement of favorite species by less desirable plants, and finally a total loss of submersed and floating-leaved plants coincident with an extension of cattail marsh. The extended marsh in turn declines, having been exposed to wave erosion through loss of the deeper zones of vegetation."

However, not all aquatic plants that increase with eutrophication are poor waterfowl foods. For example, in a study at Lake Okeechobee, Florida, Johnson and Montalbano (1984) found that waterfowl diversity in Hydrilla beds (a widespread, exotic species) was significantly greater than in several indigenous wetland plant communities (bulrush, cat-tail, pondweed, spikerush, and others).

Organic Loading/Reduced DO. The effects of severe organic loading, e.g., from wastewater outfalls, on overall community structure of wetland birds have been investigated in a few cases. Generally, abundance and/or on-site diversity of songbirds (Brightman 1976, Hanowski and Niemi 1989) and sometimes waterfowl (Belanger and Couture 1988, Piest and Sowls 1985) have tended to increase with increased abundance of aquatic invertebrates. The effect may depend on the type and configuration of the particular wastewater treatment system (Fuller and Glue 1980). Other bird groups have responded more to water levels (and associated effects on vegetation and invertebrates) than to contamination status (e.g., Ramsay 1978). In the Houghton Lake, Michigan, wetland that was exposed to treated wastewater, Kadlec (1979) reported no major shifts over a 3-year period in bird abundance or species composition.

Where introduction of organic wastes results in anoxic conditions lethal to fish and some amphibian larvae, community composition may shift from fish-eating species (e.g., herons, loons, grebes) to invertebrate-eating species and opportunists (e.g., shorebirds, songbirds, gulls, terns). Indeed, migrant shorebirds and gulls often appear to concentrate at sewage lagoons, turf farms, and wetlands mildly polluted with organic wastes (e.g., Campbell 1984, Fuller and Glue 1980).

Contaminant Toxicity. The effects of bioaccumulation of contaminants in wetland bird tissues have been widely measured, and the disasterous effects of naturally-occurring toxicants on community structure of wetlands have occasionally been documented (see discussion of Salinization below). Species assemblages for indicating the physical effects of oil spills can be easily identified based on characteristic behaviors of some wetland birds. However, the effects of pesticides, heavy metals, and other contaminants on overall structure of wetland bird communities are poorly documented in wetlands, and indicator assemblages of "most sensitive species" remain mostly speculative for these stressors (Grue et al. 1986).

Bird reproductive failure in wetlands from effects of heavy metal contamination (e.g., Scheuhammer 1987, Kraus 1989) and pesticides have been documented, but only for a few species. Lethal thresholds for metals and synthetic organics are reported in Hudson et al. (1984), EPA's "AQUIRE" database, and the US Fish and Wildlife Service's "Contaminant Hazard Reviews" series that summarizes data on arsenic, cadmium, chromium, lead, mercury, selenium, mirex, carbofuran, toxaphene, PCBs, and chlorpyrifos. However, relatively few field data are available for judging which wetland species are most sensitive. Additional testing of chemical toxicity to wildlife is currently being sponsored by EPA.

Numerous anecdotal reports exist describing relatively stable bird assemblages in traditionally-used wetlands even after years of progressive contamination. This might be attributed to the loss of nearby wetlands that otherwise would have been preferred, to behavioral avoidance of contaminated microenvironments and foods, and/or to replacement of contaminated individuals by immigrants.

Acidification. Naturally acidic wetlands sometimes have lower densities and species richness of birds, particularly in winter, than do non-acidic wetlands (Brewer 1967, Ewert 1982). Bird use of acid mine drainage wetlands in Pennsylvania was found to be less than use of natural wetlands, probably because of physical degradation of habitat rather than inferior water quality alone (Hill 1986). Acidification has also been demonstrated to reduce reproductive success and juvenile survival of some species in wetlands (e.g., tree swallows--Blancher and McNichol 1988, black ducks--DesGranges and Hunter 1987, ring-necked ducks--McAuley and Longcore 1988). Bird responses to anthropogenic acidification, summarized by McNichol et al. (1987), are felt indirectly as a result of alteration in the dominance of various food sources and possibly, changes in physical habitat (e.g., composition and distribution of submersed macrophytes). Shifts from fish to aquatic insects in lakes and streams can cause a corresponding shift from fish-eating species to those that critically depend on aquatic invertebrates, to those that feed on aquatic invertebrates opportunistically (assuming other habitat features remain relatively constant). Wetland bird groups in each category are listed in Table 14. Strong presence of a particular feeding group relative to others might be used to suggest acidification effects, if the role of other stressors (such as others listed in this section) can be ruled out.

Table 14. Examples of Wetland Birds Categorized by Major Food Source.

Predominantly feeding on fish or amphibians (at some season or life stage, in some regions):
loons, grebes, cormorants, anhinga, some herons and egrets, terns, bald eagle, osprey, kingfishers

Aquatic invertebrate obligates (at some season or life stage, in some regions):
some herons and egrets, diving ducks, some dabbling ducks, bitterns, rails, shorebirds, yellow-headed blackbird

Aquatic invertebrate facultatives:
most dabbling ducks, swallows, marsh wrens, common yellowthroat, red-winged blackbird, many other songbirds (see Adamus 1987 for list for the Northeast)

Salinization. Breeding waterfowl in hypersaline wetlands reportedly prefer fresher portions of these wetlands, and inland wetlands that are naturally saline generally have fewer nesting waterfowl (Kantrud and Stewart 1977). However, high densities of a few species, e.g., Northern Phalarope, can occur during migration in some naturally saline wetlands. The effects of salinization on structure of wetland bird communities have not been widely studied, despite publicity given to events such as the catastrophic mortality at Kesterson National Wildlife Refuge. Assemblages of species that might be used as indicators of salinization remain speculative.

Sedimentation/Burial; Turbidity/Shade. Wetland bird species that prefer soft-bottomed wetlands can be defined, but probably with insufficient precision to warrant their use as indicators of excessive sedimentation. Sedimentation affects community structure of wetland bird communities primarily by altering the type and distribution of submersed plants, and perhaps also by affecting invertebrate food sources and interfering with feeding of birds that rely on visual cues.

Vegetation Removal. Effects of vegetation removal associated with grazing and/or fire are described by Fritzell 1975, Landin 1985, Schultz 1987, and others summarized by Kantrud (1986). "Moderate" levels of grazing and/or mowing, if occurring at a time in the season when nests are not disturbed, can increase wetland bird species richness in floodplain ponds (Landin 1985) and emergent wetlands (Nelson and Kadlec 1984). However, severe grazing, mowing, or fire at inappropriate times is detrimental (Duebbert and Frank 1984, Kantrud and Stewart 1984), and total removal of woody riparian vegetation dramatically alters species composition, density, and richness of the mammalian community (Cross 1985, Malecki and Sullivan 1987, Possardt and Dodge 1978).

Many species benefit from increased openings in dense stands of vegetation and from reduced floodplain ground cover, while others, including ground-nesting species such as Northern Harrier and Short-eared Owl (USDA Soil Conservation Service 1985), do not. As patches of open water are created in formerly continuous stands of emergent vegetation, the diversity of species using a wetland typically increases (Harris et al. 1983, Kaminski and Prince 1981). These may be species that are generally widespread in the region, so the contribution of vegetation removal to overall regional diversity of birds may be slight. Species assemblages associated with vegetation structural changes can be defined by region and wetland type. Brown et al. (1989), and Durham et al. (1985) have done so for vertebrates in bottomland hardwood wetlands, and Short (1983, 1989) for midwestern and Arizona wetlands.

Effects of silvicultural activities in forested wetlands have received only limited study. Birds in forested wetlands respond very strongly to changes in vertical and horizontal vegetation structure (Finch 1990, Rice et al. 1980). Because the habitat structural needs of most forested wetland birds are relatively well-known, at least qualitatively (e.g., see Durham et al. 1985, Swift et al. 1984), indicator associations could probably be easily developed that reflect bird response to different levels and types of silvicultural practices in forested wetlands. An old-growth forested floodplain wetland in South Carolina was compared by Hamel (1989) to clearcut and selectively cut portions of the same area. More species (and particularly cavity-nesters) achieved their highest densities in the old-growth habitat than in the disturbed forested wetland, and those species that did achieve higher density in the disturbed forested wetlands were widespread throughout the region. In a southwestern riparian wetland, Carothers et al. (1974) reported 46 percent fewer breeding birds where vegetation had been thinned to 25 trees per acre, as compared to a similar reference wetland with 116 trees per acre.

Thermal alteration. The effects of thermal alteration on overall community structure of birds are poorly documented in wetlands, and indicator assemblages of "most sensitive species" remain mostly speculative for this stressor. Effects of heated wastewater are mostly indirect, affecting habitat and bird distribution by prolonging ice-free conditions in northern wetlands, altering vegetation type and structure, and affecting the type and seasonal availability of food sources (e.g., Haymes and Sheehan 1982). On a regional level, species most sensitive to changes in temperature are often those occurring at the periphery of their geographic ranges. These are easily defined by local ornithologists.

Inundation/Dehydration. The response of bird community structure to water level alteration has been the subject of dozens of studies, many conducted to improve the management of waterfowl habitat. Water level alterations (either increases or decreases) can increase or decrease overall bird abundance and richness, depending on their duration and many other factors.

Both sustained increases and sustained decreases in water levels directly affect habitat availability and dramatically shift community composition. For example, construction of dams on the lower Colorado River produced a relatively stable environment that favored high invertebrate densities and consequently increases in diving ducks, but diminished numbers of riparian species (Anderson and Ohmart 1988). Alteration of the flooding regime of a southern forested wetland from seasonal flooding to permanent flooding (for a greentree reservoir) had little overall effect on bird diversity; waterfowl and common grackles increased while white-throated sparrow and a few other species decreased (Newling 1981).

Water level changes of short durations (weeks or months), while having less affect on habitat availability, have the potential for long-term impacts to habitat quality by altering water chemistry, invertebrate populations, and seed germination. For example, dam-induced alterations in hydrologic regime have decreased bird richness partly by encouraging the spread of non-native salt cedar (Tamarix spp.)(Ohmart et al. 1977, Hunter et al. 1985).

Addition of permanent open water to a non-permanently flooded wetland usually increases the opportunity for use by waterfowl and fish-eating birds. Moreover, the typical increase in submersed and floating-leaved plants that accompanies creation of a permanent pool within a wetland provides for a more diversified plant and invertebrate food source. This consequently can result in an increase in on-site species richness of birds. Many studies have found that productivity and diversity of waterbirds are greatest within basins having a permanently flooded pool or channel that is surrounded by shallowly flooded (<10 inches depth) wetlands that are gradually dehydrated at regular seasonal or frequent (3-5 year) annual intervals (Fredrickson and Taylor 1982, Reid 1985). Among a series of Massachusetts forested wetlands, Swift et al. (1984) found that the driest wetlands supported the lowest abundance and richness of birds, even though in some regions such wetlands have the greatest diversity of vertical habitat structure and plant species richness.

Wetland bird species vary in their water depth requirements and sensitivity to water level change. Much information on depth requirements is summarized in Fredrickson and Taylor (1982) and Fredrickson and Reid (1986). This information could be used to define hydroperiod "response guilds" of birds. The most sensitive species may be those which (a) nest along water edges (e.g., Western Grebe, Redhead--Wolf (1955), or (b) feed on mudflats (e.g., shorebirds), or (c) require a particular combination of wetland hydroperiod types in a region (e.g., Kantrud and Stewart 1984). In contrast, species with nests typically well-above water level (e.g., marsh wren, prothonotary warbler) may be less vulnerable. For arid, deep-water marshes in eastern Oregon, Littlefield and Thompson (1989) suggested that presence of yellow-headed blackbird might be a good indicator of ecologically "healthy" conditions.

Fragmentation of habitat. Only a single study (Brown and Dinsmore 1986, 1988) has looked directly at the effects of fragmenting regional wetland resources. Others had previously noted the effects of fragmentation, using knowledge of species-specific life histories or data from non-wetland forest systems. Essentially, as the distance between wetlands containing certain species becomes greater, and/or hydrologic connections and vegetation corridors become severed by dehydrated channels, bank-clearing, or (particularly) roads, species most dependent on wetlands and/or which do not disperse easily could be most affected. Moreover, some species require not just a particular density of wetlands, but a particular combination of wetland types (or wetland types and other land cover types) at a particular density on the landscape or in close proximity to each other (Cowardin 1969, Kantrud and Stewart 1984, Ohmart et al. 1985, Weller 1979, Flake 1979, Patterson 1976). Although individual birds, being highly mobile, can disperse to new areas having the proper combination of types at a sufficient density, this can cause diminished reproductive success and thus, non-sustainable populations.

Territorial size requirements of wetland birds are highly variable, but can be used (with empirical observations of presence/absence in wetlands of various sizes and degrees of isolation) to define assemblages of species that are likely to be most sensitive to habitat fragmentation (Brown et al. 1989). Such studies must employ a standard level of effort (e.g., censusing time) per unit area if results are to be comparable. Radiotelemetric methods can be used to track individuals and determine home range sizes under various combinations of landscape cover patterns (Hegdal and Colvin 1986 describe techniques).

Other Human Presence. Several studies (e.g., Brooks et al. 1990, Robertson and Flood 1980, Todt 1989) have reported changes in species composition of wetland bird communities in response to general watershed "development," reduction in natural land cover types surrounding the wetlands, and increased visitation of wetlands by humans. Developed areas are characterized by a typical suite of species that include European Starling, Rock Dove, American Crow, House Sparrow, American Robin, and perhaps a few others (Graber and Graber 1976).

Human disturbance can discourage use by wildlife (Pomerantz et al. 1988), especially (a) hunting (Conroy et al. 1987, Gordan et al. 1987) and people traveling on foot (Burger 1981), and (b) during the breeding season or under harsh weather conditions. Effects of noise disturbances on wildlife are summarized by Gladwin et al. (1988). The most sensitive species appear to be ducks, geese, and other long-distance migrants which feed in large flocks at the ground or water level (Burger 1981), as well as colonially-nesting species (e.g., Markham and Brechtel 1979, Tremblay and Ellison 1979) and large species (e.g., Stalmaster and Newman 1978). Sensitivity to human disturbance may also be species-specific. Reduced use of human-visited wetlands by waterfowl or nongame waterbirds has been demonstrated by Hoy (1987), Josselyn et al. (1989), and Kaiser and Fritzell (1984). To some extent, presence of screening vegetation can permit closer approach to waterbirds by humans (e.g., Milligan 1985).

Many waterbirds take flight when humans approach within 75 to 175 feet (e.g., Josselyn et al. 1989). Wintering bald eagles may take flight when approached from a distance of 800-1,000 feet (Knight and Knight 1984; Stalmaster and Newman 1978). Motorboat activities can disturb waterfowl up to 1,000 m away (Hoy 1987). This results in more time being spent in energetically costly behaviors. Disturbance can also increase the food consumption needs of waterbirds. Korschgen et al. (1985) found that only 5 boating disturbances per day increased the energy requirements of canvasbacks by 20 percent, requiring consumption of an additional 23 g of food daily.

Other direct human influences on wetland birds include mortality from collisions with vehicles and powerlines, and predation by hunters and housecats. Hunting comprises an obvious impact to certain wetland bird species, in some cases resulting in changes at the population level.

11.2 SAMPLING METHODS AND EQUIPMENT

Some factors that could be important to measure and (if possible) standardize among wetlands when monitoring anthropogenic effects on community structure of birds include:

> distribution of water depth classes, vegetation (type, and vertical and horizontal diversity and arrangement), conductivity and baseline chemistry of waters and sediments (especially conductivity), current velocity, distance and connectedness to other wetlands of similar or different type, surrounding land cover (particularly within 500 feet of wetland perimeter), shoreline slope, wetland size, ratio of open water to vegetated wetland and its spatial interspersion, and the duration, frequency, and seasonal timing of regular inundation, as well as time elapsed since the last severe inundation or drought.

Methods for surveying bird communities are described by Burnham et al. 1980, Halvorson 1984, Ralph and Scott 1981, Verner 1985, Verner and Ritter 1985, and others. Censusing marsh and shorebirds specifically is discussed in detail by Connors (1986) and Weller (1986); censusing of waterfowl by Eng (1986) and Kirby (1980); censusing of colonial waterbirds by Speich (1986); and censusing of birds in bottomland hardwood wetlands, by Durham et al. (1985). An effort to refine techniques for monitoring wetland birds is presently being sponsored by the Maine Department of Inland Fisheries and Wildlife.

Even when apparently similar wetlands are censused, it is sometimes impossible to attribute changes in wetland bird communities to human activities within the wetland being sampled, because birds move widely across regions and continents. However, by calculating density-weighted ratios of declining resident to declining non-resident species (with similar habitat requirements), the possible role of this factor might be estimated.

Birds are present in wetlands throughout the year, but densities of birds vary greatly by season, depending on region. As with many other taxa, if only a single annual visit can be made, it should be timed to account for major life history events, such as nesting, molting, dispersal, migration, or wintering. The most severe reductions in bird density and richness occur in winter in northern emergent wetlands and bogs that completely freeze over. In southern wetlands, density and diversity are generally greatest in winter, while in northern wetlands, density and diversity are usually greatest in summer (Harris and Vickers 1984). During spring and fall, large numbers and high diversity may be present in either northern or southern wetlands. The species richness of wetlands in arid regions often increases the greatest during spring and fall, as many species seek temporary refuge during migration (e.g., songbirds in riparian oases, shorebirds in flooded fields).

A survey covering several wetlands should occur simultaneously or within consecutive days, unless severe weather conditions intervene. If the objective is to compare between-year trends in a species, total species, or species richness, then simple count methods (e.g., transects) are probably appropriate. However, if the objective is to rank wetland types or relative abundance of species, more time-consuming censusing to develop estimates of density are required (Steele et al. 1984). Determination of indices of relative annual abundance, rather than exhaustive population censusing, is suitable for most purposes (Emlen 1981).

For reasonably accurate estimates of breeding bird richness in a wetland, three visits spread over the breeding season may be desirable (Brooks et al. 1989, Weller 1986). Sampling non-wetland environments, Steele et al. (1984) reported that three repetitions of a 2 km transect were adequate to estimate bird abundance and richness of a habitat. In an inventory of birds in 87 Maine wetlands, Longcore et al. (pers.comm.) counted birds from an overview for two hours at dawn and two hours at dusk on at least two dates; as many observation points as necessary to view the entire wetland were used. The actual number and duration of visits required in a particular instance will depend on the size of the wetland, its habitat heterogeneity, visibility, and other factors.

If not only richness, but density, must be determined, then at least eight visits may be needed (Ralph and Scott 1981). Although most common songbirds will not be disturbed by frequent visits by monitoring personnel, raptors, waterfowl, other large or colonial species, and ground-nesting species may be susceptible. Wetland songbird surveys are commonly conducted in during May - July, when breeding birds are most detectable by song.

Species detection (especially of most songbirds) is greatest during early morning hours. However, thrushes and a few other species are more detectable in the evening, and in winter, some species may be most active at mid-day. Night-time coverage may be warranted, not only for typically nocturnal species such as owls, but also for waterfowl and wading birds which use different wetland types for roosting and feeding. Secretive species (e.g., rails, some passerines) have sometimes been surveyed more effectively by playing back of tape recorded calls, use of predator decoys, use of dogs, and by dragging ropes or chains through wetlands (e.g., Glahn 1974, Ralph and Scott 1981).

Surveys may be conducted from ground level, from elevated observation posts, or aerially. In the case of species that nest or roost colonially and in exposed locations, photography may be used to assist counting of individuals. Ground-level, visual techniques cannot be used effectively in wetlands with tall vegetation (mid-season emergent marshes, forested wetlands). Boats are typically used for surveys of wetlands wider than about 100 meters, as visibility from shore, even using a spotting scope, becomes restricted.

Many methods have been developed for monitoring wetland bird communities using visual, auditory, and capture techniques. These include point counts, line transects, nest counts, mist netting, and regional surveys (Brooks et al. 1989). Methods differ mainly in the degree of quantification they provide, the level-of-effort required, and the taxa they are most effective in censusing. These methods can be used in virtually all types of wetlands.

11.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

Quantitative community-level data on birds have not been uniformly collected from a series of statistically representative wetlands in any region of the country. Thus, it is currently not possible to state what are "normal" levels in wetlands of various types for parameters such as bird density, species richness, biomass, or productivity. Data on temporal and spatial variability of wetland birds among wetlands and years has been systematically collected in only a few instances. These few data sets are available largely because of the existence of two important national data collection networks, which are described as follows.

The Breeding Bird Survey (BBS) database has existed since 1966, and includes all 50 states and some Canadian provinces. Data on bird relative abundance have been collected, usually recurrently, from about

2500 transects ("routes"), each 25 miles in length and containing 50 evenly-spaced data collection points. Density of coverage varies from 1 to 16 routes per degree (latitude-longitude) block. The survey routes are not located to intentionally intersect wetlands, so wetlands are included only randomly. Routes are run only once annually, so many species may be missed. Also, some routes are conducted later in the season than is optimal for detecting some wetland species. Because routes follow roads and rely largely on auditory detection more suitable for forest birds, they may further underestimate wetland species. Nevertherless, the BBS database, by its sheer quantity of spatial and temporal coverage, represents a valuable resource for helping define "average" bird densities (in relative terms) and for aiding detection of regional trends in wetland birds. Locations of routes are included on the state maps in Appendix B.

The **Breeding Bird Census** (BBC) database also provides useful information. This database is a compilation of individual censuses conducted by volunteers throughout the United States. Compared to methods used by the BBS, the BBC protocols are more intensive, but coverage is not nearly as extensive. Whereas the BBS measures only relative abundance using a single annual visit to an area, the BBC attempts to measure population density using repeated visits. The BBC also differs from the BBS in that some habitat data are collected, but habitat heterogeneity within census plots is not quantified, the acreage of censused plots is not consistent among censuses, and only a small portion of the plots are revisited annually. In most cases, census plots are too small and heterogeneous to adequately census species with large home ranges (Terborgh 1989), as is typical in wetlands. A few of the BBC's have focused exclusively on wetlands, but these wetlands have not been chosen randomly or systematically. These are included on the state maps in Appendix B. Selected data are presented in Tables 15 and 16, located at the end of this chapter. These are based on data compilation conducted by the Cornell Laboratory of Ornithology and sponsored by the EPA Wetlands Research Program. These tables are summarized in the following paragraphs.

Median **number of breeding species** ranged from 3.5 for all censused Florida wetlands to 51 for all censused Montana wetlands, where the national maximum of 68 species was found in one censused wetland (a bulrush-cattail marsh). As expected, salt marshes at all locations had the lowest number of breeding species. The greatest variability in species richness occurred among a set of 21 Wisconsin wetlands, a set of five Kansas wetlands, and a set of seven Florida wetlands. Most repetitively-censused wetland types (NUM>1) had less than 15 percent variation in **species richness** among years, and less than 10 percent variation in **pair density** among years.

Median **density** of breeding birds (i.e., pairs per square kilometer) ranged from 138 in Alaskan wetlands to 1857 in North Carolina wetlands. The two highest densities of all counts were from riparian willow woodlands in California. One, with 4547 pairs and 35 species, was dominated by Chesnut-backed Chickadee, Bewick's Wren, Song Sparrow, and Yellow Warbler. The investigator attributed the high density to extreme density of vegetation and abundant food, despite low plant diversity. The other remarkable California riparian count, 3208 pairs per km^2 and 13 species, was dominated by Mourning Dove, Lazuli Bunting, Bewick's Wren, and Wilson's Warbler. Other high densities were in a California lacustrine marsh (3684 pairs, mainly Tricolored Blackbird), and in a cattail bulrush wetland in North Dakota (3418 pairs, mainly Yellow-headed Blackbird). The greatest variability of pair density among censused wetland types occurred among a set of four Nebraska wetlands (114 percent).

Table 15 summarizes the same parameters for each state/province, but does so **by individual years** of census. With regard to **number of species**, during a given year most states had less than 38 percent variability among their wetlands. Within any single year, the greatest variability in species richness among censused wetland types occurred between 2 Florida wetlands in 1983, which differed by 110 percent. With regard to **pair density**, during a given year most states had less than 54 percent variability among their wetlands. The greatest variability of pair density among censused wetland types occurred among 3 Colorado wetlands in 1973, which differed by 144 percent.

In general, analysis of these 478 census plots showed the following statistically significant ($p<0.05$), linear relationships, based on log-transformed data:

- the median number of species was correlated with pair density and number of repeat censuses (years) on a plot;
- variability in number of species was inversely correlated with number of species;
- the median pair density was not correlated with number of repeat censuses (years) on a plot;
- variability in pair density was correlated with pair density and number of repeat censuses (years) conducted on a plot;
- variability in pair density was correlated with variability in number of species.

Despite their statistical significance, there was considerable scatter in all of these relationships, and the correlation coefficients (r) never exceeded 0.5.

Published studies (other than from the national databases described above) that have compared year-to-year or long-term variation in bird community structure in wetlands include Bellrose 1979, Blake et al. 1987, Harris et al. 1983, Hanowski and Niemi 1987, and Rice et al. 1980. Conceivably some unpublished data on annual variation in wetland bird communities may be available from sites of the U.S. Fish and Wildlife Service's Northern Prairie Research Station, the U.S. Department of Energy's National Environmental Research Park system, and the Illinois Pool 19 and Illinois-Mississippi Rivers sites of the National Science Foundation's Long Term Ecological Research (LTER) program.

Other national bird databases exist, and new ones are being developed, for example:

- International Shorebird Survey
- Christmas Bird Count database
- Colonial Wading Bird database
- Monitoring Avian Productivity (MAP) database
- Winter Bird-population Censuses
- Migratory Waterfowl Surveys
- Mid-winter Waterfowl Survey
- breeding bird atlases in dozens of states

None of these pertain exclusively to wetlands, and it is not always possible to separate the portion of the data that includes wetlands. Still, on a collective basis, these databases could be analyzed to yield more information on community structure in different regions and occasionally, in different wetland types. Overviews of some are provided by Muir and Davis (1989) and Terborgh (1989).

Lists of breeding wetland birds have been compiled by "block" (a unit generally smaller than about 50 sq. mi.) by statewide atlas projects in many states, and along with data from Christmas Bird Counts, other databases listed above, and records kept by thousands of volunteers, these can be used to define "expected" species in wetlands. Species that show highest affinity for wetlands of various types might be identified in discussions with local birders and by accessing the "Vertebrate Characterization Abstracts" database managed by The Nature Conservancy and various state Natural Heritage Programs. Limited qualitative information may be available by wetland type from the "community profile" publication series of the USFWS (Appendix C).

Quantitative data are most available for harvested groups, like waterfowl, and least available for the majority of wetland species, which are not harvested. In a survey of waterfowl migration/ wintering habitat in the United States, Bellrose and Trudeau (1988) reported the following to represent at least "moderate" densities of waterfowl (number of birds per acre per day):

	Dabbling Ducks	Bay Divers	Geese
Atlantic Flyway	0.17	0.36	0.26
Mississippi Flyway	0.44	0.06	0.13
Central Flyway	0.73	0.09	0.34
Pacific Flyway	2.87	0.21	0.41

Of studies that have compared bird community structure among many wetlands in a region (spatial variation), perhaps the most notable for their large sample sizes are those of bottomland hardwoods by Durham et al. 1985, and prairie potholes (Kantrud and Stewart 1984, Stewart and Kantrud 1973). The latter study--of 1321 wetlands--reported the following mean densities:

Wetland class	Density (pairs/km^2)	Density (pairs/ wetland)	N
Ephemeral	200	1	4
Temporary	633.1	.76	190
Seasonal	431.8	3.52	808
Semipermanent	723.8	39.92	168
Permanent	38.6	30.80	14
Alkali	52.1	33.59	8
Fen	673.5	37.12	11
Undifferentiated tillage	89.3	0.09	118

Other quantitative studies of multiple wetlands include:

> Anderson and Ohmart 1985, Blake et al. 1987, Brewster et al. 1976, Briggs 1982, Brooks et al. 1987, 1989, Brown and Dinsmore 1986, DesGranges and Darveau 1985, Evans and Kerbs 1977, Flake et al. 1977, Hardin 1975, Harris and Vickers 1984, Heitmeyer and Vohs 1984, Hepp 1987, Hill 1986, Hudson 1983, Hunter et al. 1985, Klett et al. 1988, Knopf 1985, Landin 1985, Lawrence et al. 1985, Mack and Flake 1980, Maki et al. 1980, Menzel et al., Milligan 1985, Ohmart et al. 1985, Rector et al. 1979, Rice et al. 1980, Smith 1953, Stauffer and Best 1980, Swift et al. 1984, Wheeler and Marsh 1979, and others.

In summary, quantitative data on community composition of wetland birds is most available for breeding populations and least for wintering and migrating populations. Perhaps least-studied are montane wetlands; Northwestern wetlands; southeastern and southwestern herbaceous wetlands; and southern Great Plains wetlands. Information on impacts is most available for hydrologic alteration, vegetation removal, and acidification. Apparently the least information is available on impacts to community structure from eutrophication, sedimentation, contamination, and habitat fragmentation.

Tables 15 and 16 (Introduction).

Purpose:

These data are presented to quantitatively illustrate the variability that exists within a particular resource (wetlands). Data in this table might be used, with caution paid to the several limitations described below, to place data from a newly studied wetland into a context of other wetland studies described here. The summary metrics used here--richness and density--are only two of many community metrics that might be used for such purposes.

Explanation:

Data shown in these tables were collected by volunteers with diverse capabilities and without use of a strictly standardized protocol. Lack of standardization of study area size, and inclusion of species whose home ranges are often larger than the 10-20 mean size of most of these wetlands, introduces a significant bias into the data set. Each record below consists of a single breeding bird census, involving multiple visits during the breeding season of a single year, sometime during the period 1937-1988. The number of visits per season and the size of the censused areas varies greatly among these reported data. In Table 15, records where "NUM" >1 are sites that were visited during multiple, usually contiguous, years ("NUM" is the number of years visited). In Table 16, records where "NUM">1 are years where more than one wetland subtype in a state was visited.

These particular records were selected by the Cornell Laboratory of Ornithology as being ones most likely to include wetlands and riparian areas. This table does not contain ALL breeding bird censuses conducted in U.S. wetlands. Conversely, some records in this table may be from predominantly non-wetland habitats, or from nest colonies where densities may be atypical of overall wetland habitat. Phrases (in the "SUBTYPES" column) used to describe the sites were assigned by individual volunteers familiar with the sites, and no standardized wetland classification scheme was used. Detailed information on vegetative composition and bird species composition of most sites is available in the journal American Birds. Other columns of the tables are defined as follows:

Table 15:

MED_SPP: The median number of species, for all years when the same site was censused; when the site was censused only one year (NUM=1), the median is the cumulative total of all species found that year.

MIN_SPP: The minimum number of species in any year; when the site was censused only one year (NUM=1), the minimum is the cumulative total species found from all visits that year.

MAX_SPP: The maximum number of species in any year; when the site was censused only one year (NUM=1), the maximum is the cumulative total species found from all visits that year.

CV_SPP: The among-year coefficient of variation for all years when the same site was censused; when the site was censused only one year (NUM=1), no CV was calculated.

MED_DEN, MIN_DEN, MAX_DEN, CV_DEN: Same as above, but applying to density (number of breeding pairs per km^2).

Table 16:

MED_SPP: The median number of species, for all wetlands in the state (NUM) that were censused during the named year; years in which only one site was censused were excluded.

MIN_SPP: The minimum number of species; for all wetlands in the state that were censused during the named year.

MAX_SPP: The maximum number of species; for all wetlands in the state that were censused during the named year.

CV_SPP: The among-wetland coefficient of variation for all wetlands when several were censused the same year.

MED_DEN, MIN_DEN, MAX_DEN, CV_DEN: Same as above, but applying to density (number of breeding pairs per square kilometer).

Table 15. Within-Year Variability of Breeding Bird Richness and Density, Among Wetlands, by State.

STATE=AK

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1975	2	12.5000	12	13	5.6569	151.000	143	159	7.4925
1976	3	13.0000	7	15	35.6857	167.000	148	235	24.9502
1977	4	10.0000	8	14	23.9677	113.500	109	153	16.9409
1978	4	11.0000	11	13	8.6957	110.000	93	155	22.7207
1979	4	15.0000	8	17	31.0691	163.000	126	171	12.9728
1980	4	12.5000	10	15	19.0438	126.000	99	153	24.2897
1982	6	6.0000	5	8	18.9575	74.500	38	170	52.4533

STATE=AR

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1960	3	26.0000	23	28	9.8050	741	677	751	5.55321
1961	3	25.0000	21	27	12.5550	753	741	842	7.08589
1982	2	28.5000	28	29	2.4811	1064	1006	1122	7.70906

STATE=AZ

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1981	2	30.5000	25	36	25.5022	1053.50	410	1697	86.3831
1985	2	30.5000	28	33	11.5919	1214.00	874	1554	39.6073

STATE=CA

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1963	2	15.0000	8	22	65.9966	410.00	311	509	34.148
1972	3	27.0000	24	35	19.8357	1016.00	625	4547	104.736
1973	5	25.0000	21	30	15.2247	984.00	656	2071	46.920
1974	4	23.5000	4	28	54.5920	871.50	627	1134	32.256
1976	5	24.0000	5	31	55.2780	1165.00	314	2158	59.814
1977	13	13.0000	3	37	64.2231	1285.00	70	3208	84.863
1978	5	21.0000	6	40	64.1949	460.00	34	1107	74.003
1979	9	21.0000	4	41	62.7495	356.00	213	3843	126.900
1980	8	11.0000	3	34	75.3229	633.50	124	2134	84.092
1981	10	16.0000	2	34	57.0209	804.50	214	2305	76.483
1982	10	29.0000	3	50	62.2603	1293.50	273	2311	58.263
1983	11	37.0000	5	48	40.5472	1540.00	120	3684	58.957
1984	2	40.5000	36	45	15.7135	1271.50	1106	1437	18.408
1985	3	34.0000	25	46	30.1019	1408.00	1147	1532	14.425
1988	3	36.0000	20	40	33.0719	1097.00	1051	5600	101.182

WITHIN-YEAR VARIABILITY AMONG WETLANDS, BY STATE

STATE=CO

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1973	3	15.0000	12	25	39.2703	90.00	84	1411	144.685
1974	4	18.5000	4	24	59.5314	334.00	137	1565	113.411
1975	5	23.0000	6	24	43.5677	367.00	127	1431	95.628
1976	5	25.0000	8	30	37.7494	391.00	208	1436	86.197
1977	4	26.0000	3	34	61.1164	315.00	296	1776	108.620
1978	4	22.5000	4	37	63.6037	276.50	165	1848	125.902
1980	2	14.5000	13	16	14.6298	1232.50	776	1689	52.380
1982	2	12.0000	7	17	58.9256	1482.00	843	2121	60.977
1983	4	15.0000	6	22	50.5224	1521.00	635	2718	55.665
1984	4	13.5000	6	23	62.5425	1642.00	280	3213	71.649
1985	3	7.0000	7	14	43.3013	780.00	275	3212	110.405
1986	2	13.5000	7	20	68.0918	2968.00	2044	3892	44.027

STATE=CT

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1965	2	26.0000	20	32	32.6357	1092.50	580	1605	66.342
1968	2	25.0000	22	28	16.9706	1352.50	667	2038	71.678
1969	3	22.0000	21	34	28.1851	618.00	282	2070	95.987
1971	2	29.0000	21	37	39.0128	1472.50	981	1964	47.204
1973	2	28.0000	24	32	20.2031	1481.00	967	1995	49.082
1974	2	27.0000	22	32	26.1891	1393.00	815	1971	58.680
1975	2	26.0000	22	30	21.7571	1327.50	851	1804	50.763
1976	2	28.0000	22	34	30.3046	1556.50	900	2213	59.649
1977	2	26.0000	22	30	21.7571	1644.50	886	2403	65.228
1978	2	25.0000	24	26	5.6569	1524.50	646	2403	81.495
1979	2	19.0000	15	23	29.7729	580.50	247	914	81.247
1980	3	33.0000	33	39	9.8974	476.00	318	1934	97.973
1981	2	31.5000	30	33	6.7344	1080.00	226	1934	111.828
1982	2	24.0000	20	28	23.5702	593.00	198	988	94.201
1983	2	25.5000	23	28	13.8648	973.50	155	1792	118.904
1984	2	20.0000	10	30	70.7107	356.50	169	544	74.380

STATE=DE

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1967	2	22	20	24	12.8565	2298.00	1951	2645	21.3547
1968	2	25	24	26	5.6569	1964.50	1531	2398	31.2070

STATE=FL

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1981	3	4.0000	3	6	35.251	164	120	213	28.0819
1983	2	13.5000	3	24	109.994	206	125	287	55.6074

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                        WITHIN-YEAR VARIABILITY AMONG WETLANDS,
                                      BY STATE

---------------------------------------- STATE=IL ----------------------------------------

  DATE   NUM   MED_SPP   MIN_SPP   MAX_SPP    CV_SPP    MED_DEN   MIN_DEN   MAX_DEN    CV_DEN

  1941     2   29.5000        27        32   11.9849    681.500       600       763   16.9125
  1942     2   30.5000        26        35   20.8654    656.000       534       778   26.3009
  1944     2   25.5000        18        33   41.5945    466.500       343       590   37.4395
  1976     2    6.0000         4         8   47.1405    775.500       601       950   31.8221
  1980     2   20.5000        19        22   10.3479    505.000       451       559   15.1223

---------------------------------------- STATE=KS ----------------------------------------

  DATE   NUM   MED_SPP   MIN_SPP   MAX_SPP    CV_SPP    MED_DEN   MIN_DEN   MAX_DEN    CV_DEN

  1975     2   11.5000         4        19    92.231    788.500       297      1280   88.1529
  1978     2   18.0000         5        31   102.138    501.500       376       627   35.3906
  1979     2   20.5000         7        34    93.131    462.000       277       647   56.6298
  1983     4    8.0000         6        14    38.490    194.000        63       487   79.5463

---------------------------------------- STATE=LA ----------------------------------------

  DATE   NUM   MED_SPP   MIN_SPP   MAX_SPP    CV_SPP    MED_DEN   MIN_DEN   MAX_DEN    CV_DEN

  1975     2   27.5000        26        29   7.71389       1675      1540      1810   11.3981

---------------------------------------- STATE=MA ----------------------------------------

  DATE   NUM   MED_SPP   MIN_SPP   MAX_SPP    CV_SPP    MED_DEN   MIN_DEN   MAX_DEN    CV_DEN

  1964     2   12.0000        12        12    0.0000        696       371      1021   66.0373
  1965     2   30.5000        14        47   76.5066        589       385       793   48.9813

---------------------------------------- STATE=MD ----------------------------------------

  DATE   NUM   MED_SPP   MIN_SPP   MAX_SPP    CV_SPP    MED_DEN   MIN_DEN   MAX_DEN    CV_DEN

  1946     2   23.5000        18        29    33.099    1483.00      1235      1731   23.6497
  1947     4   29.0000        20        37    24.327    1330.00       818      1504   26.3460
  1948     9    6.0000         1        36   101.728     342.00       128      1613   93.8478
  1949     2   28.0000        25        31    15.152    1321.50      1191      1452   13.9656
  1951     2   28.5000        21        36    37.216    1409.00       919      1899   49.1813
  1971     2   34.0000        22        46    49.913    1293.00       978      1608   34.4530
  1973     2   31.5000        22        41    42.651    1171.00      1064      1278   12.9224
  1974     4   35.5000        25        44    22.254    1135.50       921      1292   13.5994
  1975     3   35.0000        26        48    30.442    1149.00       830      1206   19.0873
  1976     2   32.5000        21        44    50.041     924.00       870       978    8.2649
  1977     2   32.0000        22        42    44.194    1001.00       929      1073   10.1722
  1978     2   34.5000        24        45    43.041    1050.50      1007      1094    5.8561
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                         WITHIN-YEAR VARIABILITY AMONG WETLANDS,
                                       BY STATE

---------------------------------------- STATE=MI ----------------------------------------

 DATE     NUM    MED_SPP    MIN_SPP    MAX_SPP     CV_SPP    MED_DEN    MIN_DEN    MAX_DEN     CV_DEN

 1947       2    22.5000          7         38    97.4236        720        361       1079    70.5143
 1976       4    20.0000         16         21    12.2748        385        235        578    46.7079
 1983       2    17.0000         16         18     8.3189        293        271        315    10.6187
 1984       2    16.5000         16         17     4.2855        225        215        235     6.2854

---------------------------------------- STATE=MN ----------------------------------------

 DATE     NUM    MED_SPP    MIN_SPP    MAX_SPP     CV_SPP    MED_DEN    MIN_DEN    MAX_DEN     CV_DEN

 1979       3          3          2          5    45.8258        800        500       1050    35.1547
 1980       5          8          4         12    38.2067        293        160        800    63.2429
 1981       5          8          4         13    44.7335        473        280        816    47.5994
 1982       5          6          4         11    43.2567        300        160        648    54.9674

---------------------------------------- STATE=MT ----------------------------------------

 DATE     NUM    MED_SPP    MIN_SPP    MAX_SPP     CV_SPP    MED_DEN    MIN_DEN    MAX_DEN     CV_DEN

 1968       2    12.5000         10         15    28.2843    1478.00        699       2257    74.5380
 1975       2    41.0000         17         65    82.7832     438.50        193        684    79.1766
 1978       2    42.0000         33         51    30.3046     306.00        204        408    47.1405

---------------------------------------- STATE=NC ----------------------------------------

 DATE     NUM    MED_SPP    MIN_SPP    MAX_SPP     CV_SPP    MED_DEN    MIN_DEN    MAX_DEN     CV_DEN

 1958       2    15.5000          5         26    95.8016       1145         37       2253    136.851
 1960       2    16.5000          5         28    98.5664       1113         52       2174    134.814
 1966       2    30.5000         27         34    16.2287       1518        862       2174     61.115
 1967       2    29.0000         23         35    29.2596       1625       1037       2213     51.173
 1977       2    18.0000         11         25    54.9972        421        257        585     55.091

---------------------------------------- STATE=ND ----------------------------------------

 DATE     NUM    MED_SPP    MIN_SPP    MAX_SPP     CV_SPP    MED_DEN    MIN_DEN    MAX_DEN     CV_DEN

 1965       5     7.0000          5         10    26.7158     420.00        190        724     55.729
 1977       4    10.0000          9         13    18.2367    1876.00       1001       3418     50.599
 1978       4    17.5000         14         21    16.4957      75.00         20        389    120.619
 1979       7    14.0000          6         28    49.9300     886.00        463       1338     31.342
 1980      12    21.0000          4         30    39.2122     640.00        247       1573     58.781
 1981       2    12.0000         11         13    11.7851    2201.00       2008       2394     12.401
 1984       3    13.0000         11         16    18.8746    1030.00        907       1460     25.642
 1985       4    14.5000         12         18    18.6697    1414.50        988       1789     27.358
 1986       4    15.0000         13         18    14.5400    1503.50       1000       2263     33.308
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WITHIN-YEAR VARIABILITY AMONG WETLANDS, BY STATE

STATE=NE

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1982	2	9.5000	8	11	22.3297	221.000	198	244	14.7181
1983	2	11.0000	11	11	0.0000	284.000	256	312	13.9430
1984	2	9.5000	9	10	7.4432	205.500	195	216	7.2259
1988	2	8.0000	8	8	0.0000	260.000	240	280	10.8786

STATE=NJ

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1941	2	26.0000	10	42	87.029	492.500	326	659	47.8105
1951	2	29.0000	20	38	43.889	262.000	247	277	8.0966
1952	2	26.5000	18	35	45.362	268.000	247	289	11.0815
1953	3	24.0000	19	32	26.230	324.000	296	1097	79.4275
1954	3	21.0000	20	36	34.920	287.000	220	1065	89.6404
1956	4	19.5000	16	22	12.987	658.500	247	973	54.3934
1957	2	21.0000	20	22	6.734	624.000	383	865	54.6195
1964	2	25.5000	16	35	52.686	494.000	284	704	60.1184
1965	2	23.5000	9	38	87.260	674.500	489	860	38.8935
1966	3	17.0000	9	36	67.105	632.000	395	667	26.2056
1980	3	9.0000	4	28	92.651	877.000	303	903	48.8460
1981	3	7.0000	4	29	102.378	622.000	327	767	39.1995
1982	3	6.0000	4	29	106.865	757.000	321	767	41.4082
1983	3	11.0000	3	30	94.557	430.000	321	613	32.4533

STATE=NY

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1938	2	30.0000	28	32	9.4281	1818.50	1309	2328	39.623
1940	3	11.0000	8	18	41.6076	710.00	608	963	24.039
1941	2	16.0000	8	24	70.7107	812.00	800	824	2.090
1966	6	13.0000	5	30	57.0886	805.00	425	2250	65.854
1967	6	12.5000	5	32	60.6439	766.00	385	2808	86.035
1970	2	7.5000	5	10	47.1405	1548.50	336	2761	110.735
1982	2	30.0000	22	38	37.7124	762.50	623	902	25.873

STATE=OH

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1937	3	14.0000	13	26	40.9482	531.00	161	852	63.6963
1938	6	15.0000	5	30	52.6021	629.50	309	1284	57.1794
1939	7	17.0000	9	32	43.3085	651.00	267	1020	41.4378
1940	3	19.0000	15	23	21.0526	773.00	368	1277	56.5009
1941	3	24.0000	24	25	2.3727	865.00	316	1606	69.6854
1942	3	24.0000	23	25	4.1667	734.00	538	1729	63.8395
1944	2	14.0000	10	18	40.4061	1096.50	1015	1178	10.5115
1945	2	13.0000	9	17	43.5143	1153.50	1015	1292	16.9804
1946	3	18.0000	9	31	57.2092	1000.00	973	1275	15.4352
1947	3	17.0000	8	27	54.8330	874.00	753	1371	32.7728

WITHIN-YEAR VARIABILITY AMONG WETLANDS,
BY STATE

STATE=OH

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1948	2	12.5000	7	18	62.2254	1099.00	988	1210	14.2837
1949	2	21.5000	17	26	29.5998	980.50	672	1289	44.4962
1950	3	25.0000	20	29	18.2807	993.00	543	1146	35.0619
1955	2	27.5000	27	28	2.5713	723.50	479	968	47.7920
1956	3	26.0000	24	32	15.2317	662.00	551	1166	41.3317
1958	2	32.0000	30	34	8.8388	821.00	501	1141	55.1216
1966	3	31.0000	30	32	3.2258	981.00	669	2001	57.2436
1967	2	33.5000	33	34	2.1108	854.50	635	1074	36.3277
1974	2	35.0000	32	38	12.1218	822.00	657	987	28.3875
1975	2	36.0000	35	37	3.9284	722.50	579	866	28.0885
1977	2	31.0000	23	39	36.4958	494.00	395	593	28.3415

STATE=OK

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1940	2	19.0000	13	25	44.6594	790.500	284	1297	90.6134
1961	2	21.0000	20	22	6.7344	404.000	363	445	14.3522
1962	2	30.0000	26	34	18.8562	415.000	279	551	46.3453
1979	2	32.5000	26	39	28.2843	979.000	321	1637	95.0513
1980	2	34.0000	26	42	33.2756	653.500	347	960	66.3285
1981	2	33.0000	24	42	38.5695	792.000	326	1258	83.2100
1982	2	33.0000	27	39	25.7130	829.500	297	1362	90.7859

STATE=OR

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1983	2	12	11	13	11.7851	356.500	145	568	83.9007

STATE=PA

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1975	2	35.0000	28	42	28.2843	615.500	601	630	3.3316
1976	3	27.0000	19	40	36.9724	615.000	544	618	7.0711
1977	2	17.5000	10	25	60.6092	644.000	593	695	11.1995
1978	2	18.0000	14	22	31.4270	678.000	478	878	41.7172
1983	2	33.0000	24	42	38.5695	901.000	709	1093	30.1364
1984	2	33.0000	22	44	47.1405	809.000	577	1041	40.5559

WITHIN-YEAR VARIABILITY AMONG WETLANDS, BY STATE

STATE=SC

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1957	3	5.0000	5	5	0.0000	220.00	49	321	69.9117
1979	2	28.5000	28	29	2.4811	1558.50	1483	1634	6.8510
1980	2	23.5000	23	24	3.0090	1121.00	933	1309	23.7174
1981	6	21.0000	20	24	6.4112	1149.00	741	1476	21.4117
1982	8	27.5000	18	30	14.9288	393.00	311	1763	82.8256

STATE=UT

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1983	2	11.5000	11	12	6.14875	483.500	340	627	41.9730

STATE=VT

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1938	2	18.5000	16	21	19.111	1135.00	771	1499	45.3545
1939	2	20.5000	17	24	24.145	1397.00	904	1890	49.9075
1940	2	21.5000	17	26	29.600	1393.50	934	1853	46.6330
1941	2	24.5000	24	25	2.886	1621.50	948	2295	58.7402
1942	3	25.0000	22	29	13.863	1037.00	642	2295	65.1658
1943	3	27.0000	22	29	13.868	1052.00	692	2381	64.6998
1944	3	28.0000	21	28	15.746	1186.00	766	2559	62.3650
1945	3	29.0000	29	32	5.774	1215.00	790	3263	75.3011
1952	2	23.0000	7	39	98.380	662.00	415	909	52.7660
1953	2	21.5000	6	37	101.955	661.00	403	919	55.1993
1972	2	27.5000	20	35	38.569	584.50	426	743	38.3495

STATE=WA

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1980	2	34	33	35	4.15945	1840.50	1549	2132	22.3984

STATE=WI

DATE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
1967	2	12.5000	12	13	5.657	1035	111	1959	126.254
1969	3	8.0000	8	11	19.245	929	504	1996	67.251
1972	2	12.5000	9	16	39.598	1779	496	3062	101.992
1978	2	23.5000	8	39	93.278	1253	578	1928	76.185
1979	10	4.0000	1	33	142.349	775	350	1100	27.797
1980	2	22.0000	6	38	102.852	846	696	996	25.075
1981	5	39.0000	34	48	13.168	756	413	856	25.070

Table 16. Breeding Bird Richness and Density, by Wetland Type and State.

STATE=AK

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Moist Sedge Tussock Prostrate	1	5.0	5	5	.	68	68	68	.
Mosaic Wet/Moist Sedge Tundra	1	8.0	8	8	.	81	81	81	.
Riparian Medium Willow Thicket I	1	6.0	6	6	.	170	170	170	.
Riparian Medium Willow Thicket II	1	5.0	5	5	.	116	116	116	.
River Delta	1	11.0	11	11	.	531	531	531	.
Subarctic Taiga: Lowland	1	7.0	7	7	.	235	235	235	.
Wet Coastal Plain Tundra	3	14.0	13	19	20.9645	138	126	152	9.3842
Wet Coastal Plain Tundra I	6	12.0	10	17	21.5539	135	100	167	23.2314
Wet Coastal Plain Tundra II	6	13.5	10	17	20.4939	127	99	171	21.5570
Wet Coastal Plain Tundra III	3	11.0	8	11	17.3205	153	93	162	27.5809
Wet Coastal Tundra I	2	13.5	13	14	5.2378	154	153	155	0.9183
Wet Coastal Tundra II	1	8.0	8	8	.	116	116	116	.
Wet Sedge Tundra	1	7.0	7	7	.	64	64	64	.
Wet Sedge/Aquatic Tundra	1	6.0	6	6	.	38	38	38	.

STATE=AL

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Swamp, Upland Mixed Woods	1	32	32	32	.	909	909	909	.

STATE=AR

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Bottomland Deciduous Forest	4	20.5	17	26	17.8174	589.0	501	741	20.3138
Deciduous Creek Bottom	2	24.0	23	25	5.8926	747.0	741	753	1.1359
Deciduous Creek Bottom, Canebrake	2	27.5	27	28	2.5713	796.5	751	842	8.0787
Mature Oak Pine Bottomland	1	28.0	28	28	.	1042.0	1042	1042	.
Mississippi Delta Bottomland	2	28.5	28	29	2.4811	1064.0	1006	1122	7.7091
Oak Pine Bottomland	1	30.0	30	30	.	1215.0	1215	1215	.
Oak Pine Stream Bottomland	1	34.0	34	34	.	1662.0	1662	1662	.
Oak-Pine Stream Bottomland	1	27.0	27	27	.	966.0	966	966	.
Southern Oak Pine Bottomland	3	33.0	32	33	1.7674	1186.0	1020	1317	12.6747

STATE=AZ

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Desertscrub Riparian	1	25	25	25	.	410	410	410	.
Mesquite Hackberry Riparian	1	28	28	28	.	874	874	874	.
Riparian Woodland Live Oak	7	36	33	42	8.81808	1833	1505	2284	15.2061

BREEDING BIRD RICHNESS AND DENSITY, BY WETLAND TYPE AND STATE

STATE=CA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Annual Grassland and Vernal Pools	1	6.0	6	6	.	34.0	34	34	.
Brackish Marsh and Grassland	1	8.0	8	8	.	311.0	311	311	.
Canyon with Mountain Stream	1	29.0	29	29	.	647.0	647	647	.
Canyon; Trees and Brush on	1	32.0	32	32	.	556.0	556	556	.
Cat tail Tule Marsh	2	6.5	4	9	54.3928	2988.5	2134	3843	40.437
Cat tail Marsh with Willow Edge	1	21.0	21	21	.	2071.0	2071	2071	.
Cattail Marsh	1	12.0	12	12	.	2909.0	2909	2909	.
Coast Live Oak Riparian Woodland	1	26.0	26	26	.	915.0	915	915	.
Coastal Riparian and Cat tail	1	46.0	46	46	.	1147.0	1147	1147	.
Coastal Salt Marsh	3	3.0	2	3	21.6506	263.0	124	278	38.307
Colorado Desert Floodplain	1	14.0	14	14	.	214.0	214	214	.
Creekside Willow Woodland	1	23.0	23	23	.	1639.0	1639	1639	.
Desert Marsh	1	5.0	5	5	.	315.0	315	315	.
Desert Riparian	2	10.5	8	13	33.6718	1721.5	1285	2158	35.859
Desert Riparian Freshwater Marsh	12	38.0	34	43	8.2389	1104.5	488	1568	34.691
Desert Riparian Thicket	1	21.0	21	21	.	460.0	460	460	.
Diked Coastal Brackish Marsh	1	6.0	6	6	.	438.0	438	438	.
Diked Coastal Salt Marsh	4	10.0	5	21	62.5039	322.0	305	848	59.217
Disturbed Coastal Floodplain	1	15.0	15	15	.	772.0	772	772	.
Disturbed Coastal Salt Marsh	1	3.0	3	3	.	220.0	220	220	.
Disturbed Floodplain Woodland	3	28.0	27	30	5.3913	1134.0	1016	1156	6.832
Disturbed Riparian Stream Border	3	21.0	20	24	9.6077	1165.0	984	1322	14.619
Floodplain Riparian Forest	1	25.0	25	25	.	912.0	912	912	.
Freshwater Marsh	4	19.0	11	25	31.3638	1126.5	213	5600	121.496
Freshwater Marsh Riparian	1	31.0	31	31	.	784.0	784	784	.
High Desert Reservoir and	1	18.0	18	18	.	356.0	356	356	.
High Desert Spring	1	6.0	6	6	.	327.0	327	327	.
Lake with Marsh Edge	1	13.0	13	13	.	3684.0	3684	3684	.
Live Oaks along Canyon Creek	2	27.5	24	31	17.9991	1336.0	1104	1568	24.558
Lowland Hardwood Riparian	1	36.0	36	36	.	1051.0	1051	1051	.
Montane Meadow	1	5.0	5	5	.	120.0	120	120	.
Narrow Creekside Band of Mixed	2	43.0	41	45	6.5777	2156.5	2124	2189	2.131
Narrow Riparian Belt through	1	45.0	45	45	.	1106.0	1106	1106	.
Narrow Riverine Band of Willow	1	40.0	40	40	.	1395.0	1395	1395	.
Oak-Covered Hills; Lake	1	33.0	33	33	.	395.0	395	395	.
Open Creekside Mixed Woodland	1	40.0	40	40	.	1180.0	1180	1180	.
Restored Coastal Brackish Marsh	1	7.0	7	7	.	584.0	584	584	.
Restored Coastal Salt Marsh	2	10.0	9	11	14.1421	279.0	273	285	3.041
Riparian Hardwood Forest, Marsh,	1	36.0	36	36	.	1107.0	1107	1107	.
Riparian Marsh	1	29.0	29	29	.	829.0	829	829	.
Riparian Oak Woodland	4	24.5	23	32	15.7019	646.5	625	942	21.241
Riparian Willow Woodland	1	35.0	35	35	.	4547.0	4547	4547	.
Riparian Woodland	0	.	.	.	.	.	.	.	.
Riparian Woodland and Oak	2	31.5	29	34	11.2239	356.5	314	399	16.859
Riverine Willow Woodland with	1	37.0	37	37	.	1570.0	1570	1570	.
Riverine Woodland with Intermix	1	47.0	47	47	.	1946.0	1946	1946	.
Riverine and Creekside Riparian	2	44.0	40	48	12.8565	1722.5	1512	1933	17.283
Salacornia Salt Marsh	1	4.0	4	4	.	627.0	627	627	.
Salicornia Spartina Salt Marsh	1	3.0	3	3	.	291.0	291	291	.
Semi disturbed Riparian Woodland	1	41.0	41	41	.	752.0	752	752	.
Shrubby Meadow with Pond	2	23.5	22	25	9.0269	543.5	509	578	8.977
Suburban Park with Pond	1	12.0	12	12	.	434.0	434	434	.
Sycamore Coast Live Oak Riparian	8	25.0	17	32	20.0933	1200.5	636	1674	34.447

BREEDING BIRD RICHNESS AND DENSITY, BY WETLAND TYPE AND STATE

STATE=CA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Tamarisk Desert Riparian	1	23.0	23	23	.	106	106	106	.
Tamarisk Quailbrush Marsh	1	12.0	12	12	.	70	70	70	.
Urban Park Lake, Marsh and Mixed	2	16.0	12	20	35.3553	1633	1202	2064	37.3255
Willow Riparian	2	11.5	8	15	43.0413	1795	1285	2305	40.1810
Willow Riparian I	1	13.0	13	13	.	3208	3208	3208	.
Willow Riparian II	1	20.0	20	20	.	1829	1829	1829	.
Willow Woodland Freshwater Marsh	1	13.0	13	13	.	1476	1476	1476	.
Willow Woodland with Dense	1	36.0	36	36	.	1540	1540	1540	.
Willow Woodland with Ponded and	1	50.0	50	50	.	2311	2311	2311	.

STATE=CO

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Alpine Willow Carr, Meadow, and	2	6.5	6	7	10.8786	277.5	275	280	1.2741
Cottonwood Riverbottom Pasture	6	27.0	15	37	29.7606	172.5	84	296	43.0406
Cottonwood Willow Creekbottom	1	16.0	16	16	.	776.0	776	776	.
Floodplain Cottonwood Forest	21	23.0	6	29	44.5839	1565.0	90	3892	69.5130
Floodplain Pond	5	4.0	3	8	40.0000	518.0	165	671	45.5952
Montane Mesic Willow Carr	4	20.0	20	23	7.2289	1749.0	1409	2044	16.0672
Montane Willow Carr	2	20.5	19	22	10.3479	1521.0	1219	1823	28.0797
Plains Riverbottom	1	18.0	18	18	.	1074.0	1074	1074	.
Reservoir Shoreline Cottonwood	2	24.5	24	25	2.8862	379.0	367	391	4.4777
Semi-Wooded River Bottom Pasture	1	9.0	9	9	.	254.0	254	254	.
Semi-Wooded River-Bottom Pasture	1	14.0	14	14	.	279.0	279	279	.
Subalpine Mesic Willow Carr	1	14.0	14	14	.	780.0	780	780	.
Subalpine Willow Carr	1	11.0	11	11	.	635.0	635	635	.
Urban Cat tail Marsh and	2	16.5	16	17	4.2855	893.0	843	943	7.9183

STATE=CT

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Mature Upland Forest and Shrub	1	33	33	33	.	476.0	476	476	.
Mixed Upland Habitat and Swamp	18	30	21	39	16.3617	656.5	155	981	52.9904
Oak Forest, Thicket and Marsh	1	27	27	27	.	608.0	608	608	.
Oak and Mixed Hardwood Forest	1	32	32	32	.	580.0	580	580	.
Shrubby Swamp and Sedge Hummocks	19	22	10	33	23.5423	1934.0	544	2403	27.0926
Shrubby Swamp, Sedge Hummocks	2	23	22	24	6.1488	1933.0	1890	1976	3.1459
Upland Deciduous Forest and Marsh	1	14	14	14	.	299.0	299	299	.

STATE=DE

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Mixed Deciduous Floodplain	2	25	24	26	5.6569	1964.5	1531	2398	31.2070
Mixed Deciduous Floodplain I	1	20	20	20	.	1951.0	1951	1951	.
Mixed Deciduous Floodplain II	1	24	24	24	.	2645.0	2645	2645	.
Salt Marsh Edge	2	38	33	43	18.6081	1241.0	1203	1279	4.3304

BREEDING BIRD RICHNESS AND DENSITY, BY WETLAND TYPE AND STATE

STATE=FL

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Cordgrass Salt Marsh I	1	4	4	4	.	120	120	120	.
Cordgrass Salt Marsh II	1	6	6	6	.	164	164	164	.
Everglades Marsh	1	13	13	13	.	96	96	96	.
Gulf Coast Salt Marsh	7	3	3	3	0.00000	340	213	370	19.7072
Gulf of Mexico Barrier Island	1	24	24	24	.	125	125	125	.
Oak Palm Hickory Hammock and	4	17	15	19	9.60584	314	272	324	7.5659
Salt Marsh	1	3	3	3	.	327	327	327	.

STATE=GA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Deciduous Floodplain Forest	1	11	11	11	.	1116	1116	1116	.
Woodland Swamp	1	29	29	29	.	632	632	632	.

STATE=IA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Disturbed Creek Pasture, Pond,	1	13.0	13	13	.	662.0	662	662	.
Floodplain Tall Grass Prairie	2	11.5	11	12	6.14875	268.5	206	331	32.9193
Upland Oak-Hickory and Floodpla	1	19.0	19	19	.	289.0	289	289	.

STATE=IL

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Cat tail Marsh	1	6	6	6	.	659	659	659	.
Cattail Marsh I	1	8	8	8	.	601	601	601	.
Cattail Marsh II	1	4	4	4	.	950	950	950	.
Cut-Over Oak-Hickory with Pond,	1	32	32	32	.	763	763	763	.
Cut-Over Upland Oak-Hickory	1	35	35	35	.	778	778	778	.
Deciduous Floodplain Forest	1	34	34	34	.	637	637	637	.
Floodplain Forest	1	22	22	22	.	559	559	559	.
Grazed Stream Bottomland	1	40	40	40	.	415	415	415	.
Marsh, Meadows, etc.	1	24	24	24	.	845	845	845	.
Mixed Deciduous Second Growth	5	29	27	32	6.1789	484	454	604	11.9032
Mixed Deciduous Second growth	1	32	32	32	.	604	604	604	.
Riparian Oak Hickory Forest	1	19	19	19	.	451	451	451	.
Swamp and Thicket	1	15	15	15	.	741	741	741	.
Swampy Prairie	3	26	18	27	20.8432	534	343	600	27.1098
Upland Oak-Hickory Forest with	1	33	33	33	.	590	590	590	.

BREEDING BIRD RICHNESS AND DENSITY, BY WETLAND TYPE AND STATE

STATE=IN

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Cypress Slough	1	19.0	19	19	.	801	801	801	.
Grazed Brushy Field and Creek	1	34.0	34	34	.	803	803	803	.
Grazed Brushy Fields and Tree	2	35.5	33	38	9.9593	852	659	1045	32.0356
Grazed Brushy Fields and tree	1	39.0	39	39	.	1089	1089	1089	.
Grazed Brushy Fields, Creek	1	34.0	34	34	.	1015	1015	1015	.
Grazed, Brushy Fields and Tree	2	33.5	33	34	2.1108	925	872	978	8.1031
Hay Fields, Grazed Creek Pasture	3	39.0	38	41	3.8835	1210	1136	1356	9.0718
Hayfields, Grazed Creek Pasture	1	44.0	44	44	.	1257	1257	1257	.
Old Growth Bottomland Forest	1	48.0	48	48	.	1282	1282	1282	.
Pond and Sparsely-Wooded Meadow	1	14.0	14	14	.	496	496	496	.
Pond and Wet Meadow	4	10.5	8	13	22.6712	647	501	832	25.9168
Pond and Wet Sparsely-Wooded	1	13.0	13	13	.	973	973	973	.
Swamp Forest	1	12.0	12	12	.	350	350	350	.
Thickets and River	1	11.0	11	11	.	906	906	906	.
Two Hay Fields and Grazed Creek	1	39.0	39	39	.	1470	1470	1470	.

STATE=KS

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Brome Seeded Dikes Sewage	2	8.0	8	8	0.0000	97.0	63	131	49.5704
Floodplain Tall Grass Prairie	2	5.5	5	6	12.8565	286.5	257	316	14.5617
Floodplain Tallgrass Prairie	8	4.0	4	19	81.6967	289.0	237	1280	87.8533
Floodplain in Tall Grass Prairie	1	5.0	5	5	.	376.0	376	376	.
Riparian Woodland	1	14.0	14	14	.	487.0	487	487	.
Woodland along Stream on Prairie	2	32.5	31	34	6.5271	637.0	627	647	2.2201

STATE=LA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Floodplain Forest: Dredged	1	23	23	23	.	925	925	925	.
Impounded Bayou Gum Oak Beech	1	29	29	29	.	1810	1810	1810	.
Mature Bottomland Hardwood	1	20	20	20	.	760	760	760	.
Tupelo Swamp	1	26	26	26	.	1540	1540	1540	.

STATE=MA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Coastal Fresh water Marsh and	1	12.0	12	12	.	692.0	692	692	.
Deciduous Woodland Stream	1	14.0	14	14	.	385.0	385	385	.
Drowned Bottomland Forest	2	45.5	44	47	4.66224	711.5	630	793	16.1994
Fresh water Marsh, Swamp	1	12.0	12	12	.	1021.0	1021	1021	.
Fresh water Ponds with Mixed	1	12.0	12	12	.	371.0	371	371	.

BREEDING BIRD RICHNESS AND DENSITY,
BY WETLAND TYPE AND STATE

STATE=MA/RI

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Borders of Reservoir	1	25	25	25	.	573	573	573	.

STATE=MD

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Damp Deciduous Scrub	2	34.0	31	37	12.4784	687.0	556	818	26.9668
Deciduous Flood plain Forest	1	36.0	36	36	.	1899.0	1899	1899	.
Deciduous Floodplain Forest	6	23.5	21	28	11.0746	1577.0	1344	1712	8.0572
Floodplain Deciduous Forest	2	28.5	28	29	2.4811	1511.5	1455	1568	5.2863
Floodplain Forest with Edge	1	29.0	29	29	.	1731.0	1731	1731	.
Floodplain Forest with Openings	1	28.0	28	28	.	1504.0	1504	1504	.
Gum Beech Oak Floodplain Forest	1	36.0	36	36	.	1141.0	1141	1141	.
Hickory Oak Ash Floodplain	1	42.0	42	42	.	1073.0	1073	1073	.
Hickory Oak Ash Floodplain Fore	6	44.5	41	48	5.2345	1035.5	978	1149	7.1971
Holly Gum Oak Floodplain Forest	1	35.0	35	35	.	921.0	921	921	.
Mature Deciduous Flood Plain	1	31.0	31	31	.	1452.0	1452	1452	.
Mature Deciduous Floodplain	24	25.0	21	30	10.0780	1133.5	652	1628	27.5213
Mature Floodplain Forest	1	28.0	28	28	.	1675.0	1675	1675	.
Open Hemlock-Spruce Bog	1	25.0	25	25	.	1191.0	1191	1191	.
Second-Growth River Swamp	1	36.0	36	36	.	897.0	897	897	.
Shrubby Field with Stream-	2	19.0	18	20	7.4432	1196.5	1158	1235	4.5505
Tidal Marshes (Bulrush Saltgr)	1	5.0	5	5	.	128.0	128	128	.
Tidal Marshes (Cattail)	1	6.0	6	6	.	405.0	405	405	.
Tidal Marshes (Cordgrass)	2	1.5	1	2	47.1405	264.0	264	264	.
Tidal Marshes (Needlerush)	1	4.0	4	4	.	279.0	279	279	.
Tidal Marshes (Switchgrass)	1	10.0	10	10	.	143.0	143	143	.
Upland Mixed Forest with Small	1	35.0	35	35	.	830.0	830	830	.
Virgin Spruce-Hemlock Bog Forest	1	21.0	21	21	.	919.0	919	919	.

STATE=ME

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Beaver Pond	5	25.0	21	29	12.0363	948.0	929	1047	4.9271
Lowland Forest, Stream and Edge	2	40.5	36	45	15.7135	1047.5	860	1235	25.3141
Riparian Oak Maple Forest	1	18.0	18	18	.	1528.0	1528	1528	.

STATE=MI

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Black Spruce Tamarack Bog I	1	21.0	21	21	.	532	532	532	.
Black Spruce Tamarack Bog II	1	21.0	21	21	.	578	578	578	.
Brushy Field with Two Small	3	35.0	34	36	2.8571	326	299	361	9.4582
Brushy Field, Woodlots and Pond	6	39.0	27	44	15.8957	357	235	390	15.8976
Brushy Field, Woodlots, and Pond	1	47.0	47	47	.	363	363	363	.
Dry Meadow on Island in River	1	7.0	7	7	.	1079	1079	1079	.
Moist Deciduous Forest	2	16.5	16	17	4.2855	265	215	315	26.6833
Old Fallow Field with Pond	1	11.0	11	11	.	506	506	506	.

BREEDING BIRD RICHNESS AND DENSITY,
BY WETLAND TYPE AND STATE

STATE=MI

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Semi open Black Spruce Bog	1	19	19	19	.	238	238	238	.
Shrubby Swamp	1	41	41	41	.	2650	2650	2650	.
Tamarack Bog and Hardwood	13	17	13	20	11.6218	271	235	329	11.1135
Tamarack Swamp	1	34	34	34	.	1045	1045	1045	.

STATE=MN

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Closed Black Spruce Bog	3	11	10	12	9.0909	223	160	280	27.1606
Northern Conifer, Alder Edge,	1	18	18	18	.	682	682	682	.
Open Black Spruce Bog	3	7	5	8	22.9129	260	160	300	30.0463
Open Bog	3	6	6	8	17.3205	490	450	750	28.9155
Open Fen	3	4	4	4	0.0000	300	293	473	28.6949
Sedge Cat tail Wetland	1	5	5	5	.	1050	1050	1050	.
Sedge Calla Wetland	1	3	3	3	.	800	800	800	.
Sedge Potentilla Wetland II	1	2	2	2	.	500	500	500	.
Shrub Swamp	3	12	11	13	8.3333	800	648	816	12.2865

STATE=MT

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Alder Creek Bottom	1	5.0	5	5	.	694.0	694	694	.
Alder Creek Bottom in Montane	1	15.0	15	15	.	699.0	699	699	.
Bulrush and Cat tail Marsh in	10	53.0	39	68	18.7839	484.0	356	730	28.0226
Bulrush and Cattail Marsh in	4	65.5	52	66	11.0033	685.5	632	729	5.8161
River Falls and Western Red	1	33.0	33	33	.	204.0	204	204	.
River Floodplain Heronry	1	10.0	10	10	.	2257.0	2257	2257	.
Sage Brush Pothole Grassland	1	17.0	17	17	.	193.0	193	193	.

STATE=NC

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Cypress Pond	1	35.0	35	35	.	1037	1037	1037	.
Cypress Swamp	1	34.0	34	34	.	862	862	862	.
Flood Plain Forest	1	31.0	31	31	.	751	751	751	.
Floodplain Forest	8	25.5	20	32	17.9072	589	481	839	19.4410
Lakes John Sobotta and Claude	1	5.0	5	5	.	52	52	52	.
Mixed Maritime and Swamp Forest	1	11.0	11	11	.	410	410	410	.
Mixed Oak Pine and Swamp Forest	1	11.0	11	11	.	257	257	257	.
Mountain Lake	1	5.0	5	5	.	37	37	37	.
Woodland Swamp	18	26.5	22	28	6.7406	2174	1818	2253	5.9003

BREEDING BIRD RICHNESS AND DENSITY, BY WETLAND TYPE AND STATE

STATE=ND

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Ash Box elder Elm Hardwood Draw	2	17.5	14	21	28.2843	1314.5	1056	1573	27.8109
Ash Boxelder Elm Hardwood Draw	3	16.0	16	18	6.9282	988.0	907	1000	5.2421
Ash Cottonwood Draw	2	10.0	9	11	14.1421	985.0	868	1102	16.7983
Ash Elm Hardwood Draw	2	13.0	12	14	10.8786	1166.0	1027	1305	16.8590
Ash Elm Hardwood Draw II	2	19.5	18	21	10.8786	1356.5	1338	1375	1.9287
Ash Elm Hardwood Draw III	1	27.0	27	27	.	577.0	577	577	.
Ash Elm Hardwood Draw IV	1	21.0	21	21	.	297.0	297	297	.
Ash Hardwood Draw I	3	13.0	13	16	12.3718	1460.0	1460	1659	7.5274
Ash Hardwood Draw II	3	12.0	11	14	12.3853	1170.0	1030	1547	21.4092
Aspen Draw	3	18.0	13	19	19.2873	1789.0	510	2263	59.6306
Bromegrass wheat Grassland and	2	9.0	8	10	15.7135	201.0	190	212	7.7395
Bulrush Marsh	1	6.0	6	6	.	724.0	724	724	.
Bulrush Wetland	1	9.0	9	9	.	1582.0	1582	1582	.
Cat tail Bulrush Marsh	1	5.0	5	5	.	420.0	420	420	.
Cat tail Marsh	1	7.0	7	7	.	650.0	650	650	.
Cattail Bulrush Wetland I	1	9.0	9	9	.	1001.0	1001	1001	.
Cattail Bulrush Wetland II	1	11.0	11	11	.	3418.0	3418	3418	.
Cattail Bulrush Wetland III	1	13.0	13	13	.	2170.0	2170	2170	.
Cottonwood Floodplain I	2	28.0	28	28	0.0000	677.0	668	686	1.8800
Cottonwood Floodplain II	2	26.5	26	27	2.6683	788.5	691	886	17.4871
Cottonwood Floodplain III	1	30.0	30	30	.	594.0	594	594	.
Freshwater Lake I	1	18.0	18	18	.	89.0	89	89	.
Freshwater Lake II	1	14.0	14	14	.	61.0	61	61	.
Freshwater Wetland I	1	21.0	21	21	.	389.0	389	389	.
Freshwater Wetland II	1	17.0	17	17	.	20.0	20	20	.
Green Ash Draw	1	18.0	18	18	.	247.0	247	247	.
Hardwood Draw	1	34.0	34	34	.	688.0	688	688	.
Red Cedar Mixed Grass Prairie	2	5.0	4	6	28.2843	386.0	309	463	28.2110
Riparian Willow Shrub	1	11.0	11	11	.	2008.0	2008	2008	.
Willow Saskatoon Serviceberry	1	13.0	13	13	.	2394.0	2394	2394	.

STATE=NE

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Wetland Sedge Meadow	1	7.0	7	7	.	136	136	136	.
Wetland Sedge Meadow I	4	8.5	8	11	15.7135	250	195	280	14.6792
Wetland Sedge Meadow II	4	10.5	8	11	14.1421	228	198	312	20.7246
Wooded Creek Bottom	1	26.0	26	26	.	1304	1304	1304	.

STATE=NF

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Mature Balsam Fir Forest with	1	17	17	17	.	295	295	295	.

BREEDING BIRD RICHNESS AND DENSITY,
BY WETLAND TYPE AND STATE

STATE=NJ

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Border of Artificial Lake	1	22.0	22	22	.	872.0	872	872	.
Cat tail Bulrush Swamp	4	4.0	3	4	13.3333	694.5	613	903	18.9122
Climax Salt Marsh	1	10.0	10	10	.	659.0	659	659	.
Deciduous Forest with Pond and	20	27.5	22	31	8.8217	276.5	203	327	14.2926
Deciduous Forest, Pond, Brook	5	35.0	24	38	16.8926	667.0	324	860	34.3212
Floodplain Forest and Edge	8	15.5	10	22	22.3446	267.0	183	383	22.5870
Freshwater Bulrush Marsh with	2	7.5	6	9	28.2843	817.0	757	877	10.3859
Freshwater Marsh with Scattered	1	7.0	7	7	.	767.0	767	767	.
Hemlock Rhododendron Swamp	5	19.0	16	21	10.2316	247.0	220	324	15.2671
New Fill on Salt Marsh	1	2.0	2	2	.	7.0	7	7	.
Pine Barrens and Cedar Bog	14	35.5	30	42	10.0921	293.5	207	338	12.3352
Pine Barrens and a Small Cedar	1	34.0	34	34	.	240.0	240	240	.
Pitch Pine Barrens and Cedar Bog	2	37.0	36	38	3.8222	282.5	274	291	4.2552
Red Maple Swamp	1	11.0	11	11	.	430.0	430	430	.
Spruce, Tamarack, Maple Swamp	1	20.0	20	20	.	445.0	445	445	.
Wet Deciduous Forest	6	19.5	17	24	12.3159	919.0	632	1097	18.6893
Wet Shrubby Lowland Meadow	2	9.0	9	9	0.0000	442.0	395	489	15.0380

STATE=NM

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Mixed Mesophytic Canyon Bottom	1	35.0	35	35	.	885.0	885	885	.
Riparian	2	15.5	15	16	4.56198	510.5	430	591	22.3005

STATE=NT

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Virgin Bottomland White Spruce	1	19	19	19	.	576	576	576	.

STATE=NY

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Black Spruce Tamarack Bog	2	25.0	24	26	5.6569	1263.0	1006	1520	28.7770
Bottomland Meadow with Scattered	2	21.0	20	22	6.7344	963.5	692	1235	39.8504
Brackish Marsh	1	2.0	2	2	.	124.0	124	124	.
Brackish Marsh with Sandy Border	1	8.0	8	8	.	963.0	963	963	.
Cat tail Marsh	4	17.5	11	22	31.4949	643.5	436	727	20.4408
Cattail Marsh	1	20.0	20	20	.	647.0	647	647	.
Cattail Swamp	1	32.0	32	32	.	2328.0	2328	2328	.
Cattail Swamp, Tree Edge	1	31.0	31	31	.	2075.0	2075	2075	.
Coastal Scrub with Mosquito	2	11.5	11	12	6.1488	785.5	766	805	3.5108
Deciduous Floodplain Forest and	2	31.0	30	32	4.5620	1210.0	1136	1284	8.6489
Ditched Salt Marsh	3	5.0	5	5	0.0000	385.0	336	425	11.6691
Ditched Salt Marsh and Coastal	1	15.0	15	15	.	741.0	741	741	.
Ditched Salt Marsh and Scrub	1	13.0	13	13	.	506.0	506	506	.
Fresh water Marsh and Swamp	2	29.0	28	30	4.8766	3074.0	3004	3144	3.2204
Fresh water Marsh and Woodland	4	23.0	21	30	17.6165	2866.0	2250	3223	14.5280

BREEDING BIRD RICHNESS AND DENSITY, BY WETLAND TYPE AND STATE

STATE=NY

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Heron Colony	2	11.0	11	11	0.0000	.	.	.	.
Island in Open Lake	1	6.0	6	6	.	.	.	.	.
Lightly Wooded, Swamp Margined	1	28.0	28	28	.	1309	1309	1309	.
Marsh Bordered Island	1	25.0	25	25	.	731	731	731	.
Moist Deciduous Forest Lake	2	38.5	38	39	1.8366	905	902	908	0.4688
Phragmites and Salt-Marsh Border	2	9.5	8	11	22.3297	767	710	824	10.5098
Pond Surrounded by Orchard	1	8.0	8	8	.	40	40	40	.
Salt Marsh	1	10.0	10	10	.	2761	2761	2761	.
White Pine Red Maple Leatherleaf	1	18.0	18	18	.	309	309	309	.
Wooded Gorge with Stream and	2	21.0	18	24	20.2031	704	608	800	19.2847

STATE=OH

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Bog Forest	1	13.0	13	13	.	161.0	161	161	.
Bog and Old Filled-in Lake	1	23.0	23	23	.	538.0	538	538	.
Bogs and Ponds	1	12.0	12	12	.	971.0	971	971	.
Bur-Reed Marsh	5	9.0	7	10	13.2579	988.0	874	1015	5.9826
Disturbed Oak Hickory Forest,	21	35.0	28	39	8.0530	615.0	479	684	8.1496
Disturbed Oak Hickory, Pine	3	38.0	32	38	9.6225	586.0	466	593	13.0192
Flood-Plain Forest	2	25.5	25	26	2.7730	318.5	316	321	1.1101
Forest, Edge and Pond	5	32.0	30	36	7.3875	578.0	501	608	7.4722
Fresh Water Marsh and Adjacent	1	24.0	24	24	.	662.0	662	662	.
Grassy Marsh Deciduous Forest	2	34.5	32	37	10.2479	926.5	866	987	9.2347
Lake Shore Marsh	1	9.0	9	9	.	741.0	741	741	.
Marsh and Swamp Shrubs	3	13.0	12	14	7.6923	1020.0	741	1284	26.7522
Mature American Elm Swamp Forest	6	27.5	26	30	5.2885	1125.0	968	1267	10.1013
Nine Small Ponds	1	23.0	23	23	.	395.0	395	395	.
Oak-Hickory, Buildings and	1	32.0	32	32	.	267.0	267	267	.
Open Field and Swamp	2	14.0	13	15	10.1015	1071.0	865	1277	27.2015
Pond and Border	8	17.5	16	20	6.7389	1242.5	1146	1371	5.8838
Red Maple-Yellow Birch Bog	2	17.5	17	18	4.0406	372.0	314	430	22.0496
Shrubby River Floodplain	2	31.5	30	33	6.7344	1027.5	981	1074	6.4001
Small Bog and Field	0	.	.	.	.	852.0	852	852	.
Small Bogs and Ponds	1	20.0	20	20	.	651.0	651	651	.
Small Swamp	1	5.0	5	5	.	790.0	790	790	.
Swampy Field	2	24.5	24	25	2.8862	1667.5	1606	1729	5.2158
Upland Forest Swamp and Marsh	2	29.0	27	31	9.7532	876.5	753	1000	19.9265
Upland Hickory-Oak; Grass	1	30.0	30	30	.	309.0	309	309	.
Upland Swamp Forest and Marsh	3	25.0	23	26	6.1927	543.0	469	672	18.3018
Wet Beech-Maple	2	23.5	23	24	3.0090	819.0	773	865	7.9431
Wet Beech-Maple Woods	1	24.0	24	24	.	734.0	734	734	.
Wet Second Growth Hardwoods	1	31.0	31	31	.	2001.0	2001	2001	.
Wooded Ravine	1	17.0	17	17	.	469.0	469	469	.
Wooded Ravine and Stream	1	22.0	22	22	.	474.0	474	474	.
Wooded Ravine with Stream	1	19.0	19	19	.	368.0	368	368	.

BREEDING BIRD RICHNESS AND DENSITY,
BY WETLAND TYPE AND STATE

STATE=OK

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Bottomland Oak-Hickory-Elm	1	25.0	25	25	.	284.0	284	284	.
Bottomland-Oak-Hickory-Elm	2	29.5	28	31	7.19092	467.5	429	506	11.6465
Deciduous Floodplain Forest	7	26.0	22	28	8.14161	1362.0	960	1727	19.4846
Elm Cottonwood Oak River Border	1	21.0	21	21	.	109.0	109	109	.
Mixed Habitat Disturbed	11	42.0	39	48	7.15732	326.0	297	356	5.6569
Oak Pine Hickory Creek Bottom	1	26.0	26	26	.	279.0	279	279	.
Timbered Stream Bed and Upland	1	13.0	13	13	.	1297.0	1297	1297	.
Timbered Stream-Bed and Upland	3	12.0	11	13	8.33333	314.0	274	697	54.5207
Virgin River bottom Hardwood	1	34.0	34	34	.	551.0	551	551	.
Wooded Creek Bottom	1	32.0	32	32	.	637.0	637	637	.
Wooded Ravine, Pond and Prairie,	2	21.0	20	22	6.73435	404.0	363	445	14.3522

STATE=OR

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Sandy Beach	1	3	3	3	.	173	173	173	.
Scattered Mixed Coniferous Fore	7	12	11	18	19.3629	145	86	229	28.4486
Wet Meadow with Patchy Willows	2	13	13	13	0.0000	568	568	568	0.0000

STATE=PA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Abandoned Pasture with Stream	1	44.0	44	44	.	1041.0	1041	1041	.
Bog Mixed Hardwood Evergreen Edge	1	28.0	28	28	.	601.0	601	601	.
Bottomland Red Maple White Ash	2	19.5	15	24	32.6357	927.0	709	1145	33.2577
Deciduous Wooded Ravine, Stream	3	32.0	29	32	5.5873	815.0	635	840	14.6516
Hemlock Black Ash Swamp	1	19.0	19	19	.	618.0	618	618	.
Inland Fresh Water Marsh	2	12.0	10	14	23.5702	786.5	695	878	16.4527
Laurel Summit Glade, Mountain	1	22.0	22	22	.	577.0	577	577	.
Mixed Lowland Forest and Meadow	3	44.0	42	45	3.4981	1106.0	1093	1161	3.2230
Mountain Top Bog with Mixed	3	25.0	22	27	10.2025	544.0	478	593	10.7199
Mountain Top, Bog Mixed Hardwood	1	27.0	27	27	.	667.0	667	667	.
Stream Valley Mixed Mesophytic	2	41.0	40	42	3.4493	622.5	615	630	1.7039

STATE=RI/MA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Marsh, Swamp, and Forest During	1	31	31	31	.	1008	1008	1008	.

BREEDING BIRD RICHNESS AND DENSITY, BY WETLAND TYPE AND STATE

STATE=SC

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Hardwood Swamp Forest	4	23.0	21	29	14.8293	1659.0	1476	1835	12.3861
Oak Hickory Forest with Stream I	1	29.0	29	29	.	415.0	415	415	.
Oak Hickory Forest with Stream II	1	30.0	30	30	.	455.0	455	455	.
Swamp Hardwood Forest I	1	21.0	21	21	.	1075.0	1075	1075	.
Swamp Hardwood Forest II	1	21.0	21	21	.	1174.0	1174	1174	.
Swamp Hardwood Forest III	1	20.0	20	20	.	1297.0	1297	1297	.
Swamp Hardwood Forest IV	1	21.0	21	21	.	1124.0	1124	1124	.
Three "Carolina Bays" (Craig)	1	5.0	5	5	.	49.0	49	49	.
Three "Carolina Bays" (Ellenton)	1	5.0	5	5	.	220.0	220	220	.
Three "Carolina Bays" (Steele)	1	5.0	5	5	.	321.0	321	321	.
Upland Oak Hickory Forest with	4	27.5	25	29	6.2672	348.5	311	371	7.2395
Virgin Hardwood Bottomland Swamp	3	22.0	18	23	12.5988	741.0	729	933	14.2912
Virgin Hardwood Swamp Forest	5	24.0	23	28	8.0000	1686.0	1309	1763	11.3420

STATE=TX

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Abandoned Field with Wooded	1	10	10	10	.	628.0	628	628	.
Bottomland Hardwood Forest	1	20	20	20	.	687.0	687	687	.
Mesquite Juniper Canyon	2	18	16	20	15.7135	460.5	422	499	11.8235
Old River Bottom	1	21	21	21	.	939.0	939	939	.
Southern Floodplain Forest	1	24	24	24	.	860.0	860	860	.

STATE=UT

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Farm with Ponds	3	32	21	39	29.5884	694.0	608	1098	32.7042
Maple Box elder Creekbottom	2	13	11	15	21.7571	622.5	496	749	28.7386
Maple Boxelder Creekbottom	1	11	11	11	.	627.0	627	627	.
Riparian Woodland Hanging Gardens	2	13	12	14	10.8786	340.0	340	340	0.0000
Riparian Woodland Willow Thickets	1	26	26	26	.	491.0	491	491	.

STATE=VA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Coastal Disturbed Floodplain	2	30	24	36	28.2843	222.5	198	247	15.5722
Deciduous Floodplain Forest	1	26	26	26	.	1534.0	1534	1534	.

BREEDING BIRD RICHNESS AND DENSITY, BY WETLAND TYPE AND STATE

STATE=VT

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Alder Swamp and Pond	3	29.0	28	30	3.4483	790.0	766	827	3.8687
Balsam-Black Ash Bog	2	22.5	21	24	9.4281	837.5	771	904	11.2293
Balsam-Black Ash Bog and Border	6	28.5	25	29	6.3296	1044.5	934	1215	11.0527
Fresh-Water Pond Area	1	25.0	25	25	.	642.0	642	642	.
Northern Cutover Spruce Forest	1	20.0	20	20	.	426.0	426	426	.
Open Fresh Water Marsh	2	6.5	6	7	10.8786	409.0	403	415	2.0746
Rural Stream Border	2	34.5	34	35	2.0496	754.5	743	766	2.1555
Tamarack Bog	2	16.5	16	17	4.2855	1694.5	1499	1890	16.3162
Tamarack Bog and Border	6	22.0	17	32	21.6520	2338.0	1853	3263	19.0536
Wooded Alder Swamp and Pond	5	34.0	27	39	17.0296	820.0	692	919	13.8041

STATE=WA

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Mixed Sub climax Riparian Forest	1	33	33	33	.	2132	2132	2132	.
Red Alder Floodplain Forest	1	35	35	35	.	1549	1549	1549	.

STATE=WI

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Brushy Sedge Meadow	1	16.0	16	16	.	496.0	496	496	.
Calla Sedge Wetland	1	3.0	3	3	.	600.0	600	600	.
Cat tail Bur reed Wetland	1	2.0	2	2	.	700.0	700	700	.
Cat tail and Sedge Marsh	3	8.0	6	9	19.9242	1928.0	996	3062	51.8532
Cattail and Sedge Marsh	2	10.5	8	13	33.6718	1977.5	1959	1996	1.3230
Forested Stream Gorge I	1	38.0	38	38	.	413.0	413	413	.
Forested Stream Gorge II	1	48.0	48	48	.	655.0	655	655	.
Forested Stream Gorge III	1	39.0	39	39	.	856.0	856	856	.
Forested Stream Gorge IV	1	43.0	43	43	.	800.0	800	800	.
Inland Fresh-Water Marsh	1	10.0	10	10	.	2779.0	2779	2779	.
Lake edge Marsh	1	8.0	8	8	.	504.0	504	504	.
Mixed Wetland with Tamarack and	5	34.0	28	39	12.7709	696.0	578	756	10.2596
Sedge Cat tail Calla Wetland	1	6.0	6	6	.	950.0	950	950	.
Sedge Potentilla Cat tail	1	4.0	4	4	.	800.0	800	800	.
Sedge Bogbean Myrica Wetland	1	5.0	5	5	.	1100.0	1100	1100	.
Sedge Bur reed Wetland	1	3.0	3	3	.	750.0	750	750	.
Sedge Potentilla Wetland I	1	4.0	4	4	.	800.0	800	800	.
Sedge Wetland I	1	5.0	5	5	.	950.0	950	950	.
Sedge Wetland II	1	1.0	1	1	.	350.0	350	350	.
Spruce Tamarack Bog	1	12.0	12	12	.	111.0	111	111	.
Wet Grassy Meadow	2	10.0	9	11	14.1421	1074.5	929	1220	19.1501

BREEDING BIRD RICHNESS AND DENSITY, BY WETLAND TYPE AND STATE

STATE=WV

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Abandoned Bottomland	2	24.0	21	27	17.6777	766	593	939	31.9398
Brushy Field Bottomland	1	24.0	24	24	.	650	650	650	.
Fir Spruce Swamp	2	22.5	22	23	3.1427	762	610	914	28.2100
Mature Creek Bottom	1	26.0	26	26	.	810	810	810	.
River Bottomland Abandoned Mead	1	27.0	27	27	.	667	667	667	.
Riverside Birch Oak Forest	1	18.0	18	18	.	667	667	667	.
Spruce Laurel Bog	1	21.0	21	21	.	560	560	560	.
Wooded Marsh	1	24.0	24	24	.	469	469	469	.

STATE=WY

SUBTYPE	NUM	MED_SPP	MIN_SPP	MAX_SPP	CV_SPP	MED_DEN	MIN_DEN	MAX_DEN	CV_DEN
Western Willow-Poplar River	1	21	21	21	.	1252	1252	1252	.

12.0 WETLAND MAMMAL COMMUNITIES

12.1 USE AS INDICATORS

In general, wetlands are permanently inhabited by fewer mammal species than are upland ecosystems. However, the association of some mammals with wetlands is very strong. These include river otter, muskrat, nutria, beaver, mink, raccoon, swamp rabbit, marsh rice rat, and others. In contrast to most wetland birds, many wetland mammals are herbivores or omnivores, i.e., they consume wetland plants directly or have a mixed animal-plant diet. Muskrat in particular can have major impacts on wetland herbaceous plants (e.g., McCabe 1982). Advantages and disadvantages of using mammals as indicators are summarized in Appendix A.

As with birds, because a majority of mammals use wetlands at least briefly at some time during the year, defining what truly constitutes "wetland-dependent" is difficult. For example, individual bobcats and black and grizzly bears use wetlands extensively in some regions (e.g., Helgren and Vaughn 1989), but it is sometimes unclear whether this is the general preference of the species, and if so, whether alternative habitats infrequently visited by humans are suitable substitutes. Some species of mammals have been categorized according to wetland dependency by Brooks and Croonquist 1990, Durham et al. 1985, and Fritzell 1988.

In one comparison of existing data, prairie pothole wetlands were reported to support fewer species of mammals than either northern bogs/fens, or southern bottomland hardwoods (Fritzell 1988). Response to particular stressors is described below.

Enrichment/Eutrophication. The effects of enrichment on overall community structure of wetland mammals has not been documented, and indicator assemblages of species "most sensitive" to eutrophication remain speculative.

Organic Loading/Reduced DO. Attempts have been made in a few instances to measure the effects of severe organic loading, e.g., from wastewater outfalls, on overall community structure of wetland mammals. However, results generally have been equivocal and indicator assemblages of species "most sensitive" to organic loading remain speculative.

It can be hypothesized that, where introduction of organic wastes results in anoxic conditions lethal to mammal foods (e.g., fish and some amphibians), community composition may shift from fish-eating species (e.g., otter, mink) to vegetarian or invertebrate-eating species and opportunists (e.g., muskrat, opossum).

Contaminant Toxicity. The effects of bioaccumulation of contaminants in wetland mammal tissues have sometimes been measured. Species assemblages for indicating the physical effects of oil spills can be easily identified based on characteristic behaviors of some wetland mammals. However, the effects of pesticides, heavy metals, and other contaminants on overall structure of wetland mammal communities are poorly documented in wetlands, and indicator assemblages of "most sensitive species" remain mostly speculative for these stressors.

Acidification. Effects of acidification on the overall community structure of wetland mammals apparently have not been documented and indicator assemblages of "most sensitive" species remain speculative. It can be hypothesized that, where acidification becomes severe, community composition may shift from fish-eating species (e.g., otter, mink) to vegetarian or invertebrate-eating species and opportunists (e.g., muskrat, opossum).

Salinization. The effects of salinization, e.g., from irrigation return water and oil drilling wastes, on overall community structure of mammals has not been documented in wetlands, and indicator assemblages of "most sensitive species" remain speculative.

Sedimentation/Burial. Excessive sedimentation can alter food sources of wetland mammal communities. However, the effects of sedimentation/ burial on overall community structure of wetland mammals has not been documented, and indicator assemblages of "most sensitive" species remain speculative.

Vegetation Removal. Many mammals are sensitive to the presence and type of vegetation and its juxtapositioning with open water. Species richness of small mammals in wetlands has been correlated with complexity of vegetation structure (Arner et al. 1976, Landin 1985, Maki et al. 1980, Nordquist and Birney 1980, Stockwell 1985, Searls 1974, Simons 1985). Vegetation removal and associated long-term destruction of den sites in both wooded and emergent wetlands has resulted in changes in furbearer populations and small mammal communities (Krapu et al. 1970, Malecki and Sullivan 1987, Possardt and Dodge 1978), while restoration of riparian vegetation has led to increases in use by mink (Burgess and Bider 1980). However, many small mammals are more abundant in the denser herbaceous ground cover that results from overstory removal, as shown in a Texas riparian system by Dickson and Williamson (988). Grazing at levels recommended by the Soil Conservation Service had no significant effect on abundance or distribution pattern of small mammals in a Colorado cottonwood floodplain (Samson et al. 1988).

Species in Iowa considered by Geier and Best (1980) to be least tolerant of vegetation change include Microtus pennsylvanicus, Spermophilus tridecemlineatus, Reithrodontomys megalotis, Peromyscus maniculatus, and Mus musculus. Species considered "moderately tolerant" included Sorex cinereus and Blarina brevicauda. The Eastern chipmunk (Tamias striatus) and white-footed mouse (Peromyscus leucopus) were considered the most tolerant in Iowa, and this was also found to be true in the Vermont study of Dodge et al. (1976) and Possardt and Dodge (1978). Species considered most sensitive to riparian vegetation removal in Vermont were jumping mice (Zapus hudsonicus and Napeozapus insignis) and shrews (Blarina brevicauda, Sorex cinereus).

Geier and Best (1980) predicted that a reduction in shrub cover would reduce populations of T. striatus and S. cinereus. T. striatus would be especially affected by the selective removal of eastern red cedar. Populations of T. striatus, Peromyscus leucopus, and the two shrew species would suffer from the loss of woody plant debris (logs, brushpiles, and stumps).

Despite these initial efforts, indicator assemblages of mammals "most sensitive" to vegetation removal remain speculative in most of the U.S., and the effects of vegetation removal on overall community structure of mammals have not been well-documented in wetlands.

Thermal alteration. The effects of thermal alteration on overall community structure of mammals apparently have not been documented in wetlands, and "most-sensitive" indicator assemblages remain speculative.

Dehydration/Inundation. Changes in wetland water level and soil moisture alter the quantity and quality of mammal habitat, and may trigger immigration and emmigration of particular species. The effects of dehydration may be particularly severe if they occur during hibernation, due to the effects of exposure. In northern wetlands, muskrats, for example, require deep water in winter for successful hibernation (Bellrose and Low 1943). Although muskrats and minks appeared to tolerate temporary flooding in an Illinois forested floodplain, opossums, red foxes, gray foxes, striped skunks, and woodchucks were evicted by flood conditions (Yaeger 1949).

In northern Florida cypress ponds, Harris and Vickers (1984) found an increase in relative abundance of rice rats and a decrease in cotton rats with any addition of water. In a series of Maine bogs, species richness of small mammals was highest in the driest part of the bog, near the upland edge (Stockwell 1985). In prairie pothole wetlands, small mammals select habitats based on soil moisture levels (Pendleton 1984). In Colorado (Olson and Knopf 1988), mammal species richness, relative diversity, and faunal similarity were greater in upland communities than in riparian wetlands. Richness was also less in Washington riparian areas than in adjoining uplands, although presence of water-placed woody material within the wetter areas mediated this effect to some degree (Mason 1989).

Fossorial mammals (e.g., moles and shrews) that inhabit subsurface areas may be particularly sensitive to moisture level changes. However, local changes in moisture regimes and other aspects of wetland habitat quality are frequently not reflected by indicator species of mammals because of the ability of mammals to move freely, in and out of impacted areas.

Despite these initial efforts, indicator assemblages of mammals "most sensitive" to habitat dehydration or inundation remain speculative in most of the U.S., and the effects of these stressors on overall community structure of mammals have not been well-documented in wetlands.

Fragmentation/Isolation of Habitat. Although habitat fragmentation has been widely implicated in the decline of some large mammals, we found little explicit documentation of overall mammal community response to fragmentation of regional wetland resources. One can surmise that as the distance between wetlands containing wetland-dependent mammals becomes greater, and/or hydrologic connections and vegetated corridors become severed by dehydrated channels, bank-clearing, or (particularly) roads, the more sensitive mammals or those which do not disperse easily might be most affected. Although individual mammals, being highly mobile, can disperse to new areas having the proper combination of wetland types at a sufficient density, they probably do so at risk of greater predation and energetic loss.

Sensitive species can be grouped into "guilds" that exhibit similar responses to fragmentation. For example, Brooks et al. (1989, 1990) found significant differences in mammal communities in disturbed vs. undisturbed watersheds, and recommended that stream corridors be at least 100 m in width. Home range sizes of wetland mammals have also been used for defining wildlife guilds and required buffer strip sizes (Brown et al. 1989). However, home range sizes can vary greatly by season and habitat type. They can be determined from observations of presence/absence in wetland patches of various sizes and degrees of isolation, or by using radiotelemetry (Hegdal and Colvin 1986 describe techniques).

12.2 SAMPLING METHODS AND EQUIPMENT

Some factors that could be important to measure and (if possible) standardize among wetlands when monitoring anthropogenic effects on community structure of mammals include:

> distribution of water depth classes, vegetation and woody debris (type, and vertical and horizontal diversity and arrangement), current velocity, distance and connectedness to other wetlands of similar or different type, surrounding land cover (particularly within 500 feet of wetland perimeter), wetland size, ratio of open water to vegetated wetland and its spatial interspersion, and the duration, frequency, and seasonal timing of regular inundation, as well as time elapsed since the last severe inundation or drought.

Methods for surveying mammal communities are described in Cooperrider et al. (1986), Halvorson (1984), and others.

Mammals occur in wetlands throughout the year. Mammal density and richness may be reduced during and immediately after floods in riverine wetlands. Surveys covering several wetlands, if not conducted simultaneously, should occur within consecutive days, unless severe weather conditions intervene. For efficient censusing, advantage can be taken of species that congregate seasonally in wetlands (e.g., white-tailed deer in northern cedar swamps). Diurnally, detection of most species is greatest at night. Visual surveys of larger, day-active species can be conducted from ground level, from elevated observation posts, or aerially. Low-altitude overflights or aerial photography can be used to identify some beaver dams and beaver and muskrat lodges, and to census moose and large mammals in open country. Ground-level, direct-observation techniques cannot be used effectively in wetlands with tall vegetation (mid-season emergent marshes, forested wetlands).

Many methods have been developed for monitoring wetland mammal communities, and generally rely on various types of traps. Tracks, scat, den trees, burrows, vocalizations, eyeshine, and other sign may also be counted using point counts, line transects, or similar methods. Some species can be attracted to scent stations or salt blocks. Most non-capture methods can be used in virtually all types of wetlands. Methods differ mainly in the degree of quantification they provide, the level-of-effort required, and the taxa they are most effective in censusing. Thus, whenever possible a variety of methods should be used.

Spring-loaded snap traps, live (cage) traps, pitfall traps, and funnel traps are widely used for capturing mammals. Animals are attracted by bait or, in the case of pitfall traps, stumble into a confining pit and usually cannot escape. They are subsequently identified, counted, measured, and released. To reduce loss of trapped animals to predation, traps and funnels are checked regularly (at least every other day) and can be shaded, and/or filled with sufficient moist plant litter to minimize physiologic stress to animals.

The efficiency of traps and funnels can be increased by channeling small animal movements in the direction of the trap or funnel. This is commonly done with "drift fences" (Gibbons and Bennett 1974). These are fences constructed of wire screen or polyethylene plastic, with lengths of at least 5-15 m. Lengths less than 2.5 m are not very effective (Bury and Corn 1987). Traps are placed at both ends of the drift fence, along the fence at various points, or at the junction of several intersecting fences. The bottom edge of the fence is emplanted in the ground, or at least no space is provided for non-burrowing animals to crawl under the fence. Sizes and shapes of containers and associated drift fences and their configurations vary greatly, depending partly on target species and wetland type. Trap and funnel methods can provide relatively quantitative data, when arranged systematically and level-of-effort (e.g., "trap-hours") is standardized.

The size of the trap, baits used, and trap placement can affect the species that are caught. Thus, a variety of methods should be used if possible (Szaro et al. 1988). Snap traps are effective for cricetids and many other small rodents (e.g., meadow vole, short-tail shrew, house mouse, western harvest mouse, masked shrew)(Geier and Best 1980), whereas pitfall traps are more effective for rodents that are primarily insectivorous and/or fossorial (moles and shrews)(Szaro 1988). Funnel traps are ineffective in capturing many forest mammals (Bury and Corn 1987). If only a single type of capture method can be used and the aim is to capture the widest variety of small mammals, then in Pacific Northwest forests, Bury and Corn (1987) recommend use of pit traps over a continuous 60-day period; a list of the most common species could be compiled by using pitfall traps only for a typical 10-day trapping period. However, the high water table in many wetlands can render pitfall traps impractical due to flooding. In these situations, spring-loaded traps mounted on floating platforms are effective for detecting some species (pers. comm., T. Roberts, Waterways Experiment Station, Vicksburg, MS).

Examples of community-level mammal studies in wetlands include, for example:

> Cross 1985 (Oregon), Geier and Best 1980 (Iowa), Landin 1985 (Mississippi), McConnell and Samuel 1985 (West Virginia), Olson and Knopf 1988 (Colorado), Scelsi (n.d.)(New Jersey), and Urbanek and Klimstra 1986 (Illinois).

12.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

In general, quantitative data on structure of the entire mammalian community of wetlands has not been uniformly collected from a series of statistically representative wetlands in any region of the country. Thus, it is currently impossible to state what are "normal" levels for parameters such as mammal density, species richness, or biomass, and their temporal and spatial variability, in any type of wetland.

We found only a few published studies that quantified the entire mammalian community (or a large proportion of it) among a set of wetlands: Brooks et al. 1985, 1987, 1989, Geier and Best 1980, Landin 1985, Nordquist and Birney 1980, Pardue et al. 1975, Stockwell 1985, and Urbanek and Klimstra 1986 (Illinois).

We found no journal articles that quantified year-to-year or long-term variation in mammalian community structure in wetlands, but conceivably such unpublished data may be available from sites of the U.S. Department of Energy's National Environmental Research Park system, sites of the National Science Foundation's Long Term Ecological Research (LTER) program, and regional studies of the ELF military communications facility (Blake et al. 1987).

Quantitative data on composition of wetland mammalian communities is virtually lacking from all regions except parts of the Northeast and some riparian systems. Information on impacts is limited mostly to studies of hydrologic effects and vegetation removal; especially little is known of impacts to community structure from contaminants, salinization, sedimentation, and habitat fragmentation.

Qualitative lists of "expected" mammals in wetlands can be easily developed in most regions from Niering (1985), Fritzell (1988), and the "Vertebrate Characterization Abstracts" database managed by The Nature Conservancy and various state Natural Heritage Programs. Limited qualitative information may be available by wetland type from some of the "community profile" publications of the USFWS (Appendix C).

However, fine gradations in degree of dependency of individual species upon wetlands have not been defined. Quantitative data are most available for harvested species, while the majority of wetland mammals, which are not harvested, are seldom studied quantitatively in wetlands.

13.0 BIOLOGICAL PROCESS MEASUREMENTS IN WETLANDS

13.1 USE AS INDICATORS

This discussion addresses biological processes that are commonly monitored in inland wetlands. "Processes" here are considered to be synonymous with wetland "functions." Included are litterfall and decomposition, nutrient translocation, growth and production, and respiration. We have limited consideration mainly to studies where these processes have been monitored for an entire wetland, not just a dominant species or community within the wetland. Relatively few studies have monitored wetland biological processes in the context of evaluating a specific anthropogenic stressor, as has Mader et al. (1988).

Although an understanding of wetland processes and their vulnerability to anthropogenic stressors is fundamental for predicting future impacts, the limited evidence to date suggests that biological processes usually respond only weakly and slowly to stressors in wetlands. This may be because biological processes represent the net result of many potentially compensating mechanisms within biological communities (Schaeffer et al. 1988, Schindler 1987). In contrast to changes in community structure which tend to occur gradually, changes in processes, when they ultimately occur, may occur suddenly and catastrophically. Perhaps with further testing and development of new ways to measure and quantify biological processes, their utility to regulatory monitoring programs will increase. Advantages and disadvantages of use of wetland ecosystem processes as indicators of ecological condition are shown in Appendix A.

Enrichment/Eutrophication. The effects of enrichment on annual productivity, decomposition and denitrification have been studied primarily in cypress dome and northern bog wetlands. Responses are generally typical of what has been found in other aquatic systems--increased productivity with "moderate" enrichment and a decline in productivity with "severe" enrichment.

Effects of enrichment on decomposition rates are highly variable, with both increased decomposition and no effect reported (Almazan and Boyd 1978, Andersen 1979, Chamie 1976, Fairchild et al. 1984, Farrish and Grigal 1988, Meyer and Johnson 1983, Richardson et al. 1976). Differing conclusions may be due to differences in current velocity, leaf type, temperature, fertilizer type, ambient water quality, and other factors. Enrichment of wetlands with nitrogen-rich runoff may lead to an increased proportion of nitrous oxide release (vs. N_2 release), which is of potential concern because even small changes in the production of nitrous oxide are potentially significant considering the role of this gas in destroying stratospheric ozone (Hahn and Crutzen 1982).

Enrichment commonly increases secondary production. For example, aquatic invertebrate production was correlated with enrichment (total phosphorus concentration) in Plante and Downing's (1989) analysis of aquatic bed community data from 51 lakes (164 samples) from temperate regions of the world.

Organic Loading/Reduced DO. The effects of severe organic loading, e.g., from wastewater outfalls, on annual productivity have been studied primarily in cypress dome and northern bog wetlands, and results were similar to the above. With regard to decomposition, Brinson et al. (1981) reviewed the available literature and concluded that decomposition in wetlands should occur most rapidly with aerobic conditions under some optimum regime of wetting and drying; alternating conditions of aerobic and anaerobic result in slower decomposition.

Contaminant Toxicity. The literature summary by Baath (1989) reports heavy metal-induced impairment of several microbial processes, such as respiration, phosphatase enzyme activity, denitrification (Grant and Payne 1982) and decomposition of leaf litter (Jackson and Watson 1977), in wetland soils. In one case enrichment has been demonstrated to mitigate toxicity effects (Fairchild et al. 1984). In general, the relative toxicity of metals to microbial processes decreases in the order Cd> Cu> Zn> Pb (Baath 1989). Cadmium in shrub wetlands can interfere with nitrogen fixation (Wickliff et al. 1980). The effects of metals on primary and secondary production, and the effects of other contaminants on other processes, have not been widely studied in wetlands.

Acidification. The effects of acidification on biological processes have generally not been studied in inland wetlands. Long-term decomposition rates, particularly of the most refractile litter components, are generally slower in acidic water bodies (Friberg et al. 1980), and few kinds of decomposer bacteria operate effectively below pH 4 (Doetsch and Cook 1973). Artificial acidification has been shown to decrease the decomposition rate of litter from an herbaceous wetland plant (Leuven and Wolfs 1988), but the degree of inhibition may depend on the buffering capacity of the litter (Gallagher et al. 1987). Increasing the pH by adding lime can speed decomposition in acidic wetlands (Ivarson 1977). Acidification can also affect nitrification rates in wetlands (Dierberg and Brezonik 1982), and secondary production. Aquatic invertebrate production was correlated inversely to pH in Plante and Downing's (1989) analysis of aquatic bed community data from 51 lakes (164 samples) from temperate regions of the world.

Salinization. The effects of salinization, e.g., from irrigation return water and oil drilling wastes, on biological processes have generally not been studied in inland wetlands.

Sedimentation/Burial. The effects of excessive sedimentation on biological processes have generally not been studied in inland wetlands. Based on studies in other surface waters, respiration is likely to increase initially and decomposition rates may decrease.

Turbidity/Shade, Vegetation Removal. The impacts of increased turbidity on biological processes have generally not been studied in inland wetlands. Based on studies in other surface waters, primary production increases with increased solar energy, and secondary production may increase as well, depending on habitat availability and other factors. Decomposition in a southern forested wetland, as measured by the "cotton rate of rotting (CRR)" was 50 percent greater after removal of vegetation by herbicide than in an undisturbed forest (Mader et al. 1988).

Thermal Alteration. Decomposition may be enhanced by moderate temperature increases, but thermal effects are more likely to be overshadowed by effects of litter type, depth, consumer invertebrate density, and canopy cover (Hauer et al. 1986). Primary and secondary production generally increase with increasing temperature, but thresholds beyond which these processes start to decline are not known for any wetland type, and thermal loading may decrease the primary productivity of specific taxa and communities (e.g., Scott et al. 1985). Aquatic invertebrate production was correlated with water temperature in Plante and Downing's (1989) analysis of aquatic bed community data from 51 lakes (164 samples) from temperate regions of the world.

Dehydration/Inundation. In southern floodplains, production of woody vegetation was greater in forested wetlands that are flooded during some portion of the year but are well-drained (except for small, intermittent storms) during the growing season (Birch and Cooley 1983). Decomposition rates are generally slower in wetlands with longer duration flooding, anoxia, and greater water depths (Brinson 1981, Day et al. 1988), but dehydrated wetlands may experience considerable accretion of organic matter (Burton 1984, Elder and Cairns 1982). Effects of various inundation regimes on vegetation biomass have been reported by Knighton (1985), Fredrickson and Taylor (1982), Robel 1962, and others.

Fragmentation of Habitat. We found no studies that attributed a decline in individual wetland annual productivity, decomposition or denitrification rates to the regional declines in wetlands that have occurred. One can surmise that as the distance between wetlands becomes greater, and/or hydrologic connections become severed by dehydrated channels or dams, the simplified community structure of the remaining wetlands would support lower biological rates. However, this has not been tested.

13.2 SAMPLING METHODS AND EQUIPMENT

Some factors that could be important to measure and (if possible) standardize among wetlands when monitoring anthropogenic effects on the processes of annual productivity, decomposition and denitrification include:

> age of wetland (successional status), water depth, temperature (site elevation, aspect), hydraulic residence time, conductivity and baseline chemistry of waters and sediments (especially pH, DO, organic carbon, and suspended sediment), current velocity, sediment type, stream order or ratio of discharge to watershed size (riverine wetlands), shade, ratio of open water to vegetated wetland, vegetation type, and the duration, frequency, and seasonal timing of regular inundation, as well as time elapsed since the last severe inundation or drought.

Methods for measuring productivity and other biological processes in aquatic environments are described in Aloi 1990, Edmondson and Winberg 1971, Kibby et al. 1980, Murkin and Murkin 1989, Smith and Kadlec 1985, Symbula and Day 1988, and others. A method for measuring whole-wetland respiration is described by Madenjian et al. (1990).

Because all biological processes are expressed as rates, they require data from at least two points in time. To measure annual productivity in wetlands, measurements of plant biomass are made at the onset of the growing season and at the time of peak live biomass. Measurements of decomposition are generally initiated during the mid to late growing season.

Methods that have been used in wetlands are described only briefly below. Measurement of biological processes in wetlands has generally been done with great innovation and adaptation, with few studies employing exactly the same procedures. Thus, only two measurements are described below--decomposition and tree growth. For other processes and parameters, methods used in other surface waters, e.g., for measurement of invertebrate production, might sometimes be applicable to wetlands.

Decomposition methods. Typically, several packs of biodegradable material are placed in surface water and subsets are removed over periods ranging from weeks (usually) to years. Decomposition is inferred by difference in weight over a specified period of time.

Organic matter decomposition rate in one wetland study was measured by as the tensile strength losses of soil burial cloth (93 percent cellulose) after 9 days (Mader et al. 1988). In another study, cypress leaves in mesh fiberglass screen bags were placed in the deepest spots (Dierberg and Ewel 1984); these authors cited the finding of Deghi et al. (1980) that there is no significant difference in decomposition rates between center and edges of cypress swamps. Five litter bags were collected at 15, 29, 58, 114, 205, 390, and 570 days. In a third study (in a stream), bags of air-dried leaves collected just before leaf-fall were placed in riffles in a control and a treatment stream. Bags were collected at 10, 30, 58, 87, and 115 days after placement (Meyer and Johnson 1983).

The litter decomposition rate can integrate short-term indexes of microbial activity (such as ATP, CO2 evolution, and microfaunal counts) over periods of several years (Edmonds 1987). Tree leaf and grass litter is collected and air dried; litter bags are set out and collected at 1, 2, 5, and 7 years.

Decomposition of different sections of three plant species was studied by Hill (1985), who collected Nelumbo leaf laminae and petioles, Typha leaves, and whole Ludwigia plants in the fall when the leaves were beginning to turn yellow. The litter was air-dried and cut into 10-cm pieces, and 2-5 g samples were put into nylon mesh leaf bags (15 cm^2, 3-mm octagonal openings). Three to five replicates of each type were put between wire mesh to hold them on the sediment below the water level of a reservoir. Samples were collected from an inundated site at 2, 4, 7, 14, 21, 28, 63, 91, 119, and 154 days and from a drawdown site at 2, 4, 7, 14, 35, and 63 days. Macroinvertebrates were removed from the samples before air-drying.

Tree Growth. Increment cores can be used to estimate tree ages and growth rates, as well as for shrubs (Ehrenfeld 1986). Data from ring counts can be checked against aerial photographs (Klimas 1987). Lemlich and Ewel (1984) took cores of pondcypress (Taxodium distichum var. nutans), a difficult species to age because of the presence of false rings. They identified false rings by their gradual change in cell size, as contrasted with true rings, in which small latewood cells are readily distinguishable from large earlywood cells.

Leavitt and Long (1989), working with southwestern conifers, described a method of using tree ring analysis to reconstruct historic precipitation and drought patterns. Their method is based on ratios of ^{13}C to ^{12}C, using the principle that, under drought conditions when stomates are closed, the tree will use a greater proportion of carbon-13 in photosynthesis.

Repeated measurement of tree diameter also can be used to gauge growth. It is important to define precisely where on the trunk the measurement is to be taken. In Franklin and Frenkel's study (1987), tree data could not be compared between years because the heights on the boles at which diameters were measured were not standardized. Straub (1984) took diameters of cypress trees at 1.37 m above ground or above buttresses, if present. Small nails were hammered into the trunks so remeasurement would be done at the same point on the tree. Aluminum vernier tree bands calibrated to one-hundredth of an inch are also used to measure tree growth (Sklar and Conner 1983).

More detailed measurements of diameter were used by Scott et al. (1985) in a South Carolina floodplain swamp. Tree biomass was measured by taking five diameters at 5-cm intervals above and below breast height. A nail was driven into the stem at the topmost measuring point to facilitate subsequent measurements; a chain with measurement intervals marked on it can be hung from the nail.

13.3 SPATIAL AND TEMPORAL VARIABILITY, DATA GAPS

In general, data on community-level biological processes have not been uniformly collected from a series of statistically representative wetlands in any region of the country. Thus, it is currently impossible to state, for any wetland type, what are "normal" rates for processes such as annual productivity, decomposition and denitrification.

Only a few studies have compared biological processes among wetlands or aquatic environments in a region or among regions. These include Brinson et al. 1981, Cushing et al. 1983, and Plante and Downing (1989).

Apparently few studies have compared year-to-year or long-term variation in biological processes in wetlands. Such unpublished data may be available from sites of the U.S. Department of Energy's National Environmental Research Park system, and sites of the National Science Foundation's Long Term Ecological Research (LTER) program.

Existing data on wetland plant productivity, collected by a wide variety of methods, was reported by Adamus 1983, Kibby et al. 1980, and (for Carex wetlands only) by Bernard et al. 1988. Net annual primary productivity of some inland wetland emergent species can exceed 6000 $g/m^2/yr$, but usually is less than about 2000 $g/m^2/yr$. Biomass of submersed macrophytes spans four orders of magnitude (Moeller 1975). Decomposition of emergent macrophytes in lacustrine wetlands may take from about 200 to 1000 days for 90 percent weight loss (Hill 1985). Breakdown rates (per day) range from 0.0008 for woody plants in bogs to 0.0190 for non-woody plants in riparian wetlands (Webster and Benfield 1986).

Secondary production in wetlands has been measured much less often than primary production. For invertebrates, Smock et al. (1985) reported 3.09 g/m^2 annual production from an acidic South Carolina forested wetland; Plante and Downing (1989) compile estimates of invertebrate production from lacustrine wetlands. Fish production in a 4-year study of the Okefenokee Swamp in Georgia ranged from 43 to 187 kg wet mass/ha (Freeman 1989).

Limited quantitative data on other biological processes is available by wetland type in some of the "community profile" publications of the USFWS (Appendix C).

14.0 LITERATURE CITED

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APPENDIX A. Summary of Advantages and Disadvantages of Use of Major Taxa in Monitoring Wetland Ecological Condition.

Microbial Communities

ADVANTAGES

- tight linkage to fundamental processes (e.g., decomposition, denitrification, respiration)
- samples easily collected, transported, and analyzed
- some taxa linked to animal welfare (e.g., streptococci)
- EPA protocols available
- immediate response to contamination
- measurable in wetlands which lack surface water
- sensitive to presence of some contaminants (e.g., Ames test, Microtox test)
- "indicator taxa" relatively well-known (especially protozoans)
- some culture bioassay data are available

DISADVANTAGES

- response is often not identifiably stressor-specific
- laborious and slow (plate culture) identification; process measurements difficult to interpret with regard to ecological significance
- general absence of existing regional field databases
- rapid turnover requires frequent sampling; do not integrate conditions over time very well
- naturally great micro-spatial variation, especially in tidal wetlands
- drifting cells in riverine wetlands complicate interpretation
- low social recognition of their importance
- bioaccumulation is irrelevant and impractical to detect

Algae

ADVANTAGES

- tight linkage to fundamental processes (e.g., photosynthesis, respiration)
- pivotal relationships in food webs
- EPA protocols available (may need modification to wetlands)
- measurable in some wetlands which lack surface water
- tolerances and indicator value are relatively well-known, particularly to nutrients, and most are very sensitive to herbicides
- simple collection procedures with minimal wetland impact
- response to stressors is usually immediate
- generally immobile and thus reflective of site conditions, useful for in situ exposure assessments and whole-effluent bioassays

DISADVANTAGES

- response is often not identifiably stressor-specific
- laborious identification
- some regional field databases exist, but not for wetlands
- rapid turnover requires frequent sampling
- cannot be effectively sampled during dormant season
- low social recognition of their importance
- bioaccumulation is unmeasurable
- drifting cells of unattached species complicate interpretation

- most relatively insensitive to heavy metals and pesticides (Hellawell 1986)

Mosses, Liverworts, Ferns

ADVANTAGES
- a few taxa are reputed indicator species for physicochemical contaminants
- perhaps the most sensitive indicator of hydric regimes
- the only integrator of the long-term geologic record (ie, peat core analyses for metals accumulation, land cover change, ground water flow reversals)
- immobile and thus reflective of site conditions, useful for in situ exposure assessments

DISADVANTAGES
- response is often not identifiably stressor-specific
- laborious sampling and identification
- low social recognition of their importance
- few regional field databases exist

Submersed Aquatic Vascular Plants

ADVANTAGES
- extremely sensitive to turbidity, eutrophication, hydroperiod
- sensitivities of several indicator species are well known
- relatively important in food webs (e.g., waterfowl)
- immobile and thus reflective of site conditions, useful for in situ exposure assessments
- patterns interpretable using remote sensing

DISADVANTAGES
- difficult to sample systematically throughout a wetland
- cannot be effectively sampled during dormant season
- absent from wetlands that lack standing water (e.g., bogs)
- tolerant of intermittent pollution
- laborious identification
- low social recognition of their importance
- few regional field databases exist

Non-rooted Aquatic Vascular Plants

ADVANTAGES
- extremely sensitive to nutrient additions
- sensitivities of some indicator species (e.g., Lemna) are well known
- important in food webs (e.g., waterfowl)
- mostly immobile and thus reflective of site conditions, useful for in situ exposure assessments
- patterns sometimes interpretable using remote sensing

DISADVANTAGES
- difficult to sample systematically throughout a wetland
- limited bioaccumulation due to short lifespan
- absent from wetlands that lack standing water (e.g., bogs)
- laborious identification

- low social recognition of their importance
- few regional field databases exist
- cannot be effectively sampled during dormant season

Emergent (Herbaceous) Vascular Plants

ADVANTAGES
- occur in virtually all wetlands
- sensitivities of some indicator species (e.g., Typha, Phragmites, Phalaris) to nutrients/sediment are well known
- immobile and thus reflective of site conditions, useful for in situ exposure assessments
- bioaccumulate to a moderate degree
- patterns interpretable using remote sensing
- sampling techniques and community metrics well-developed
- moderately sensitive to nutrients and hydroperiod alteration
- some regional field databases exist

DISADVANTAGES
- not highly sensitive to contaminants and sedimentation
- lagged response to stressors (episodic contamination may not be reflected)
- low social recognition of importance
- sampling and identification is laborious
- community cannot be completely characterized during the dormant season
- dispersal, herbivory, soil type and other factors often overshadow contaminant effects

Forested/Shrub (Woody) Vascular Plants

ADVANTAGES
- occur widely
- sensitivities of many species to hydroperiod change are relatively well known
- immobile and thus reflective of site conditions
- bioaccumulate to a moderate degree
- patterns interpretable using remote sensing
- sampling techniques and community metrics well-developed
- some regional field databases exist
- trends can be inferred (with care) using tree ring analyses
- signs of stress (e.g., die-offs) are socially recognized
- sampling and identification are fairly easy
- community can be characterized even in the dormant season

DISADVANTAGES
- not highly reflective of contaminants and sedimentation
- long lagged response to stressors (episodic contamination may not be reflected); in situ experimentation is impractical
- response difficult to interpret where past management (e.g., silviculture) has been practiced

Aquatic Insects (e.g., dragonflies, midges)

ADVANTAGES

- occur in all wetland types, even those lacking surface water
- community metrics/indices well-developed (e.g., Index of Biotic Integrity) but may need adaptation for wetlands
- intermediate lifespans reflect episodic events without requiring extremely frequent sampling
- bioaccumulate to a moderate degree
- can be caged for whole-effluent bioassays or in situ assessments
- relatively important in food webs
- community can usually be sampled year-round
- some regional field databases exist, though few for wetlands
- show characteristic response to all major wetland stressors (hydroperiod, sediment, nutrients, contaminants)
- some taxa linked to human welfare (e.g., mosquitoes)
- EPA sampling protocols available, but need modification for wetlands
- contaminants may induce identifiable deformities

DISADVANTAGES
- occurrence in isolated wetlands may be strongly tied to sources of colonizers and their dispersal mechanisms
- sampling difficult and true densities very difficult to determine in wetlands with herbaceous vegetation
- laborious identification
- low social recognition of their importance
- naturally great micro-spatial variation
- community composition potentially affected by selective predation (e.g., by fish, waterfowl)

Benthic/Epiphytic Macro-crustaceans (e.g., amphipods, crayfish, oligochaetes, isopods)

ADVANTAGES
- less subject to dispersal than aquatic insects (and thus more reflective of conditions in a particular wetland)
- may be more sensitive than aquatic insects to contaminants
- fairly simple sampling and identification
- social recognition of some species (e.g., crayfish, sandworms)
- other advantages — similar to Aquatic Insects, above

DISADVANTAGES
- mostly absent from wetlands which lack standing water
- naturally great micro-spatial variation
- community composition potentially affected by selective predation (e.g., by fish, waterfowl)

Mollusks

ADVANTAGES
- highly immobile and thus most reflective of site conditions, useful for in situ exposure assessments
- highly bioaccumulative (e.g., clams, mussels)
- depuration procedures can indicate potential contaminant uptake rates
- bioassay data fairly extensive
- contaminants may induce identifiable deformities
- can be sampled year-round
- historic recreation of growth is possible (with care)

- presumptive indicator of hydroperiod (complete, sustained wetland drawdown)
- EPA protocols available
- high social importance of coastal species (shellfish)

DISADVANTAGES
- very localized occurrence, related largely to dissolved solids rather than contaminants
- laborious sampling and (in freshwater) identification

Fish

ADVANTAGES
- community metrics well-developed (Index of Biotic Integrity), though not for wetlands; many reputed indicators (e.g., carp)
- most comprehensive set of bioassay data
- can be caged for whole effluent bioassay and in situ studies, or avoidance measured using radiotelemetry
- moderately bioaccumulative
- fairly simple identification (except larval stages)
- population characteristics, growth fairly easy to discern
- contaminants may induce identifiable deformities
- can be sampled year-round
- presumptive indicator of hydroperiod (absent from isolated wetlands with complete, sustained drawdown)
- EPA protocols available
- integrate broad, longer-term, landscape-level impacts because of their mobility, high trophic position, and longer life span
- high social importance of most species; existing water quality standards for aquatic life focus on fish

DISADVANTAGES
- mobility makes it difficult to locate specific contaminant sources
- absent (or present for only brief periods) in most wetlands
- laborious sampling
- early life stages and non-game species may be difficult to identify

Amphibians and Reptiles

ADVANTAGES
- small home range relative to larger vertebrates
- highly (e.g., snapping turtle, alligator) to moderately bioaccumulative; can be caged for in situ assessments
- some social recognition
- fairly simple identification
- fairly well-established sampling protocols
- sensitive to hydroperiod alteration
- present in most inland wetland types

DISADVANTAGES
- sampling limited to certain seasons in some regions
- mostly absent from tidal wetlands
- sampling can be laborious

- presence can be strongly influenced by natural dispersal conditions

Birds

ADVANTAGES
- high social recognition, particularly waterfowl
- have the only relatively extensive nationwide databases on trends, habitat needs, distribution
- moderately extensive bioassay data
- some species (e.g., wading birds, harrier) are highly bioaccumulative
- avoidance is measurable using radiotelemetry, and in situ assessments are possible (caged or clipped individuals)
- simple sampling and identification
- present in all wetland types
- established sampling protocols are available
- the only suitable indicator of degradation occurring at the landscape scale

DISADVANTAGES
- in general, community structure is highly controlled by physical habitat, and perhaps hunting mortality, rather than contaminants
- mobility makes it difficult to locate specific causes of mortality sources (could be thousands of miles away)
- essentially absent from some wetlands in winter

Mammals

ADVANTAGES
- many (e.g., otter) are highly bioaccumulative
- high social recognition and value (e.g., muskrat)
- avoidance is measurable using radiotelemetry, and in situ assessments are possible (caged individuals)
- fairly simple sampling and identification
- some sign (e.g., beaver dams) can be remotely sensed
- present in all wetland types
- established sampling protocols are available
- an extensive database of acute toxicity data for mice/rats may be partially transferable

DISADVANTAGES
- great temporal and spatial variation (many species are cyclic) makes data interpretation difficult
- in general, community structure is highly controlled by physical habitat, and perhaps trapping mortality, rather than contaminants
- mobility (and frequent use of non-wetland habitat) makes it difficult to locate specific causes of mortality sources

Biological Processes (Functions)

Definition: Whole-wetland measurement of photosynthesis, primary productivity, respiration, denitrification, nitrogen fixation, decomposition, leaching, and/or similar processes

ADVANTAGES
- most important indicators of wetland sustainability and life support function

DISADVANTAGES

- not as sensitive to contamination as is community structure or tissue analysis (Schindler 1987)
- measurement is laborious, time-consuming (e.g., isotopes)
- social recognition of importance is weak
- extreme spatial and temporal variation
- measured values may reflect natural successional stage rather than human-induced stress

APPENDIX B. Wetland Biomonitoring Sites, Referenced and Mapped by State.

The following maps are provided (a) to facilitate regionalization of future efforts, (b) to further cooperation among researchers, and (c) to encourage use/analysis of extant data. **These maps and their associated bibliographies DO NOT depict ALL wetland research sites.** Nonetheless, a systematic, extensive process was used to develop them, as described in section 1.2. Criteria for including studies in this listing were described in section 1.2, but a small portion of sites may not fully meet these criteria. The quality of individual studies or descriptions of their locations have not be verified or assured. Digital versions of all maps and their bibliographies currently reside with the Wetlands Team at EPA's Environmental Research Laboratory in Corvallis, Oregon.

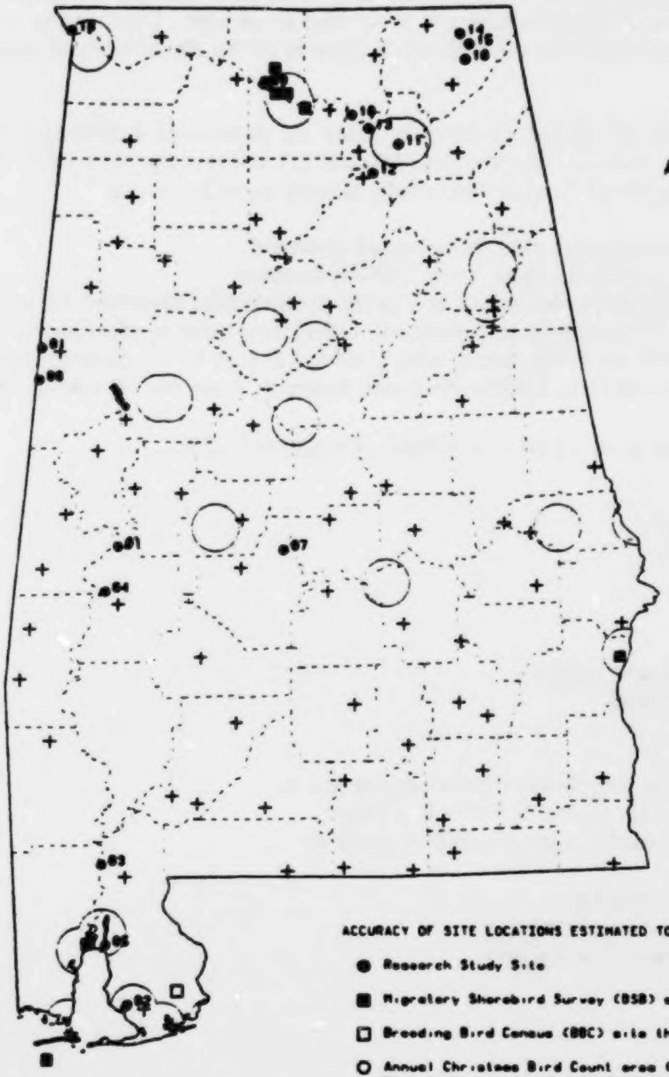
The numbers on the maps are keyed to citations listed on pages that follow each state map. The abbreviation on the line above each citation (e.g., MOBBC21) includes the state code, number reference to map, and (in some cases) the following additionally identifying abbreviations:

BBC = wetland breeding bird census plot, from Cornell database
BBS = wetland breeding bird survey route, from USFWS database
BSB = inland shorebird migration site, monitored by the International Shorebird Survey
BW = waterfowl survey site (breeding, mid-winter, or other) monitored by state and/or federal agencies
LTR = long-term environmental research site, usually funded in part by the National Science Foundation
EPA = reference wetland studied by USEPA Wetlands Research Team and its contractors

Abbreviations at the end of citations refer to topical coverage, as follows:

A = algae
AI= aquatic invertebrates
B = birds
BA= bioaccumulation
D = decomposition
F = fish
H = herptiles (amphibians and reptiles)
I = impacts of human activities
MA= mammals
MI= microbial communities
P = plants (generally), PB= bog plants, PE= emergent plants,
    PM= submersed macrophytes, PW= woody plants
R = regional studies (> 5 wetlands simultaneously sampled)
RS= remote sensing
S = spatial distribution of wetlands
SO= sediment/organic matter accumulation
TS= time series measurements (> 3 years)

Inland Wetlands Having Biological Community Measurements



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi.

- ● Research Study Site
- ■ Migratory Shorebird Survey (BSB) site
- ☐ Breeding Bird Census (BBC) site that includes wetland
- ○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat
- + Breeding Bird Survey Starting points for 25mi. transects AND points where transects enter new county Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

- ◆ State/Federal waterfowl survey

This map does NOT portray ALL wetland sampling sites Emphasis is on sites where community-level data were collected See chapter 1 for inclusion criteria

Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Bentoria Cartography Jeff Irish July 1988

ALABAMA

Mapped

AL1
Pennington, C.H., J.A. Baker, F.G. Howell, and C.L. Bond. 1981. Study of Cutoff Bendways on the Tombigbee River. Tech. Rep. E-81-14, U.S. Army Engr. Waterw. Expt.Stn., Vicksburg, MS. F

AL3
Mader, S.F., W.M. Aust, and R. Lea. 1988. Changes in Net Primary Productivity and Cellulose Decomposition Rates in a Water Tupelo-Bald Cypress Swamp Following Timber Harvest. Fifth Biennial S. Silvicul. Res. Confer., Memphis, TN. 5 pp. PW D I

AL4
Teels, B.M., G. Anding, D.H. Arner, E.D. Norwood and W.E. Wesley. 1978. Aquatic plant-invertebrate and waterfowl associations in Mississippi. Proc. Southeast. Assoc. Fish Wildl. Agenc. 30:610-616.

AL5
Aust, W.M., S.F. Mader, and R. Lea. 1988. Abiotic changes of a tupelo-cypress swamp following helicopter and rubber-tired skidder timber harvest. Fifth Southern Silviculture Res. Conf., Memphis, TN. PW I

AL7
Hall, T.F. and W.T. Penfound. 1943. Cypress-gum communities in the Blue Girth Swamp near Selma, Alabama. Ecol. 24(2):208-217. PW

AL7
Hall, T.F., W.T. Penfound, and A.D. Hess. 1946. Water level relationships of plants in the Tennessee Valley with particular reference to malaria control. J. Tenn. Acad. Sci. 21:18-59.

AL8
Auburn University. 1982. Fisheries studies on Gainesville and Aliceville Lakes on the upper Tombigbee River system, Alabama-Mississippi.Environ. Studies Inland Waterway Sys., CESAM-PD-EI. F

AL9-12
U.S. Fish & Wildl. Service. 1985. Survey of waterfowl utilization activities, Tennessee-Tombigbee Waterway, Alabama and Mississippi. USFWS Division of Ecological Services, Daphine, AL. B

AL9-12
U.S. Fish & Wildl. Service. 1986. Tennessee-Tombigbee Waterway, waterfowl survey. 1985-1986 annual Report. Division of Ecological Services, Daphine, AL. B

ALABAMA (continued)

AL9-12
U.S. Fish & Wildl. Service. 1987. Tennessee-Tombigbee Waterway, waterfowl survey. 1986-1987 annual Report. Division of Ecological Services, Daphine, AL. B

AL9-12
U.S. Fish & Wildl. Service. 1986. A study of cutoff bendways on the Tennessee-Tombigbee Waterway. 1986 Annual Report. Division of Ecological Services, Daphine, AL. AI F

AL13
Pardue, W.J. and D.H. Webb. 1985. A comparison of aquatic macroinvertebrates occurring in association with Eurasian Watermilfoil with those found in the open littoral zone. J. Freshw. Ecol. 3(1):69-79.

AL14
Tomljanovich, D.A., G.A. Brodie, and D.A. Hammer. 1988. Constructed Wetlands for Treating Acid Drainage at TVA Facilities. Off. of Nat. Res. and Economic Dev., TVA/ONRED/WRF-88/2. Knoxville, TN.

AL15-16
James, W.K., D.R. Lowery, D.H. Webb, and W.B. Wrenn. 1989. Supplement to White Amur Project Report. Tennessee Valley Authority. Resource development, River Basin Operations, Water Resources. TVA/WR/AB--89/1. Muscle Shoals, AL.

AL18
Peltier, W.H. and E.B. Welch. 1969. Factors affecting growth of rooted aquatic plants in a river. Weed Sci. 17:412-416. P PM

AL18
Peltier, W.H. and E.B. Welch. 1970. Factors affecting growth of rooted aquatic plants in a reservoir. Weed Sci. 18:7-9. PM I

ALBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

ALBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

ALBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

ALBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

ALABAMA (continued)

ALCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Deutsch, W.G. 1988. Community structure and production of benthic macroinvertebrates in 0.04-hectare ponds with and without organic loading. Ph.D. Diss., Auburn Univ., Auburn, AL. 163 pp.

Johnson, R.C, J.W. Preacher, J.R. Gwaltney, Jr., and J.E. Kennamer. 1975. Evaluation of habitat manipulation for ducks in an Alabama beaver pond complex. Southeastern Assoc. Game Fish Comm. Proc. 29: 512-518.

Miranda, L.E., W.L. Shelton, and T.D. Bryce. 1984. Effects of water level manipulation on abundance, mortality, and growth of young-of-the-year largemouth bass in West Point Reservoir, Alabama - Georgia. N. Amer. J. Fish. Manage. 4:314-320. F

Murad, H.A. 1987. Acidification as environmental pollution: effects on fishpond ecology. Ph.D. Diss., Auburn Univ., Auburn, AL. 101 pp.

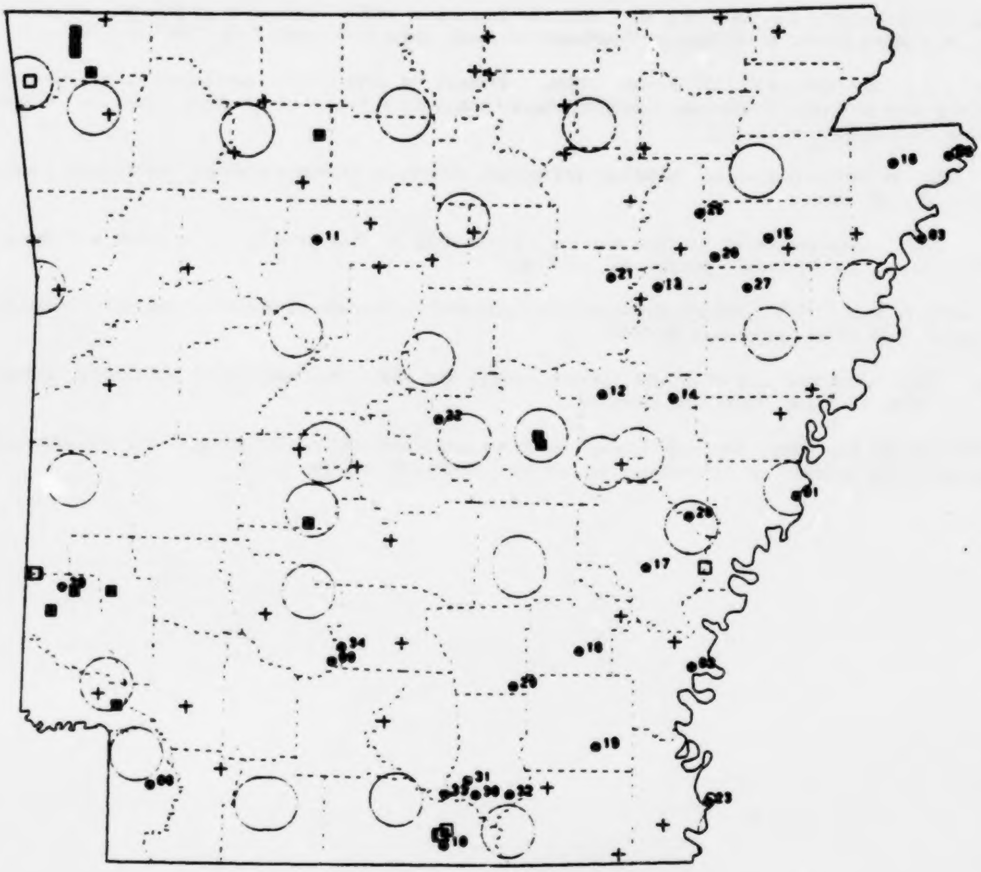
Sedana, I.P. 1987. Development of benthos and its relationship to fish production in ponds with organic loading. Ph.D. Diss., Auburn Univ., Auburn, AL. 118 pp.

Smith, B.M. and E.P. Hill. 1979. The potential of coal strip mine in Alabama as waterfowl habitat. Proc. Ann. Conf. S.E. Assoc. Fish Wildl. Agencies: 33:1-10. B

Speake, D.W. 1955. Waterfowl use of creeks, beaver swamps, and small impoundments in Lee County, Alabama. pp. 178-185 In: Proc. SE Assoc. Game Fish Comm. B

Webb, D.H. and A.L. Bates. 1989. The aquatic vascular flora and plant communities along rivers and reservoirs of the Tennessee River system. J. Tennessee Acad. of Sci. 64(3):197-203. PM

Inland Wetlands Having Biological Community Measurements



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SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

- ◆ State/Federal waterfowl survey

Arkansas

This map does NOT portray ALL wetland sampling sites Emphasis is on sites where community-level data were collected See chapter 1 for inclusion criteria

Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Rentoria Cartography Jeff Irish July 1988

ARKANSAS

Mapped

AR1-3
Landin, M.C. 1985. Bird and Mammal Use of Selected Lower Mississippi River Borrow Pits. Ph.D. Diss., Mississippi State Univ., 405 pp. B MA

AR4-7
Cobb, S.P., C.H. Pennington, J.A. Baker, and J.E. Scott. 1984. Fishery and ecological investigations of main stem levee borrow pits along the lower Mississippi River. Mississippi R. Comm., Vicksburg, MS. 120 pp. F

AR6
Rainwater, W.C. and A. Houser. 1982. Species composition and biomass of fish in selected coves in Beaver Lake, Arkansas, during the first 18 years of impoundment, 1963-1980. N. Amer. J. Fish. Manage. 2(4):316-325.

AR8-20
Dale, E.E.,Jr. 1984. Wetland forest communities as indicators of flooding potential in backwater areas of river bottomlands. Arkansas Water Resour. Res. Center Pub. #106, Univ. of Arkansas, Fayetteville, AR. Proj. G-829-08. PW

AR21
Harris, J.L., F.L. Burnside, B.L. Richardson, and W.K. Welch. 1984. Methods for analysis of highway construction impacts on a wetland ecosystem--a multidisciplinary approach. pp. 7-17 In: Wetlands and Roadside Management. Transportation Res. Rec. 969, Trans. Res. Bd., National Research Council, Washington, D.C.

AR21
Harris, J.L., F.L. Burnside, B.L. Richardson, and W.K. Welch. In Prep. Eight-year biomonitoring of Oats Creek forested floodplain wetland, Bradford, Arkansas.Arkansas State Highway and Transportation Dept., Little Rock, AK.

AR22
Bryant, C.T., C.T. Bryant, Jr., and J.D. Rickett. 1988. Use Attainability Analysis of Rosenbaum Lake Pulaski County, Arkansas. Water Res. Assoc. of Arkansas, Little Rock. F

AR23
Baker, J.A., C.H. Pennington, C.R. Bingham, and L.E. Winfield. 1987. An Ecological Evaluation of Five Secondary Channel Habitats in the Lower Mississippi River. U.S. Army Corps of Engr., Mississippi River Comm., Lower Mississippi River Environ. Prog., Rep. 7. Vicksburg, MS.

AR25-34
State of Arkansas, Dept. of Pollution Control and Ecology. 1987. Physical, Chemical, and Biological Characteristics of Least-Disturbed Reference Streams in Arkansas' Ecoregions, Vol. I - Data Compilation. 685 pp. F

ARBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

ARBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

ARBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

ARBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

ARCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

ARKANSAS (continued)

Not Mapped

Bedinger, M.S. 1971. Forest Species as Indicators of Flooding in the Lower White River Valley, Arkansas. Prof. Pap. 750-C, U.S. Geol. Surv., Reston, VA.

Bedinger, M.S. 1979. Forests and Flooding with Special Reference to the White River and Ouachita River Basins, Arkansas. Rep. 79-68, U.S. Geol. Surv., Reston, VA. 27 pp. PW

Cobb, S.P. and A.D. Magoun. 1985. Physical and Hydrologic Characteristics of Aquatic Habitat Associated with Dike Systems in the Lower Mississippi River, River Mile 320 to 610, AMP. U.S. Army Corps of Engr., Mississippi River Commission, Lower Mississippi River Environ. Prog., Rep. 5. Vicksburg, MS.

Huffman, R.T. 1980. The relation of flood timing and duration to variation on bottomland hardwood community structure in the Ouachita River Basin of Southeastern Arkansas. U.S. Army Engr. Waterways Exp. Stn. Miss. Paper E-80-4, Vicksburg, MS. 22 pp.

Hupp, C.R. and E.E. Morris. 1990. A dendrogeomorphic approach to measurement of sedimentation in a forested wetland, Black Swamp, Arkansas. Wetlands 10:107-124.

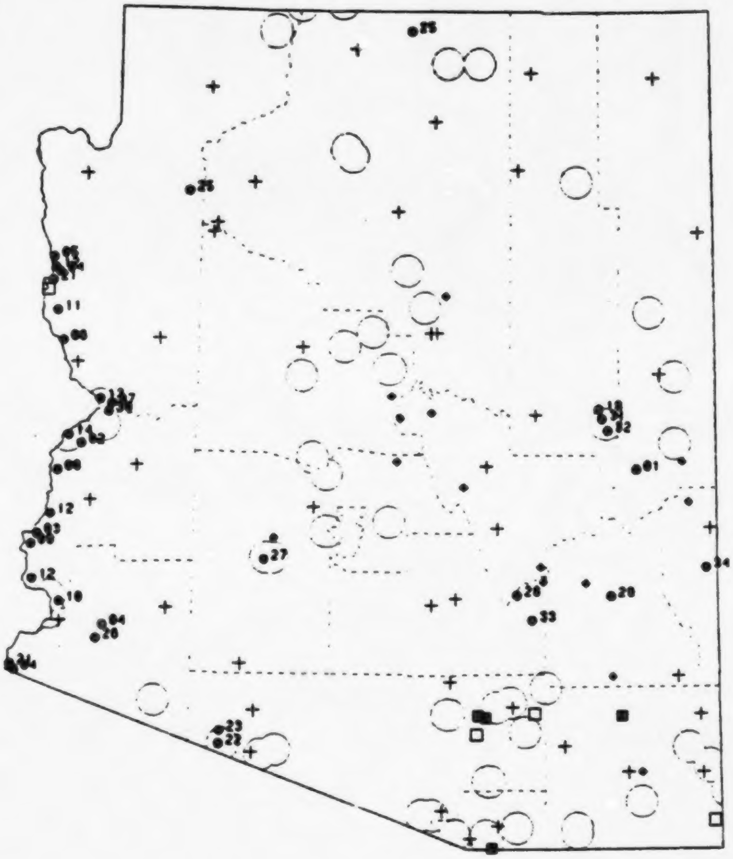
Klimas, C.V. 1988. Forest Vegetation of the Leveed Floodplain of the Lower Mississippi River. U.S. Army Corps of Engr., Mississippi River Commission, Lower Mississippi River Environ. Prog., Rep. 11. Vicksburg, MS.

Klimas, C.V., C.O. Martin, and J.W. Teaford. 1981. Impacts of Flooding Regime Modification on Wildlife Habitats of Bottomland Hardwood Forests in the Lower Mississippi Valley. U.S. Army Engr. Waterw. Expt.Stn., Rep. # EL-81-13. 200 pp. I

Lowery, D.R., M.P. Taylor, R.L. Warden, and F.H. Taylor. 1987. Fish and Benthic Communities of Eight Lower Mississippi River Floodplain Lakes. U.S. Army Corps of Engr., Mississippi River Commission, Lower Mississippi River Environ. Prog. Rep. 6. Vicksburg, MS. 299 pp.

Schramm, H.L., Jr. and C.H. Pennington. 1981. Aquatic habitat studies on the lower Mississippi River, River Mile 480 to 530. Rep. 6., Environ. Lab. U.S. Army Engr., Waterw. Expt.Stn., Vicksburg, MS. Misc. Paper E-80-1. 74 pp. F

Inland Wetlands Having Biological Community Measurements



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SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

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Arizona

This map does NOT portray ALL wetland sampling sites Emphasis is on sites where community-level data were collected See chapter 1 for inclusion criteria

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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

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ARIZONA

Mapped

AZ1
Heede, B.H. 1985. Interactions Between Streamside Vegetation and Stream Dynamics. pp. 54-58 In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. PW

AZ2-3, 11
Ohmart, R.D., B.W. Anderson, and W.C. Hunter. 1985. Influence of agriculture on waterbird, wader, and shorebird use along the lower Colorado River. pp. 123-127 In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. I B

AZ4
Ohmart, R.D., B.W. Anderson, and W.C. Hunter. 1988. The Ecology of the Lower Colorado River from Davis Dam to the Mexico-United States International Boundary: A Community Profile. U.S. Fish & Wildl. Serv. Biol. Rep. 85(7.19). 296 pp. B P

AZ5-10
Radtke, D.B., W.G. Kepner, and R.J. Effertz. 1988. Reconnaissance Investigation of Water Quality, Bottom Sediment, and Biota Associated with Irrigation Drainage in the Lower Colorado River Valley, Arizona, California, and Nevada, 1986-87. U.S. Geol. Surv. AI BA I

AZ8
Rice, J., B.W. Anderson, and R.D. Ohmart. 1980. Seasonal habitat selection by birds in the lower Colorado River Valley. Ecol. 6(6):1402-1411. B R

AZ10
Jones, K.B. and P.C. Glinski. 1985. Microhabitats of lizards in a southwestern riparian community. pp. 342-346 In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. H

AZ10-12
Anderson, B.W. and R.D. Ohmart. 1988. Structure of the winter duck community of the lower Colorado River: Patterns and processes. pp. 191-193 In: M.W. Weller (ed.). Waterfowl in Winter. Univ. Minnesota Press, Minneapolis. B

AZ10-12
Meents, J.K., B.W. Anderson, and R.D. Ohmart. 1984. Sensitivity of riparian birds to habitat loss. pp. 619-625 In: R.E. Warner and K.M. Hendrix (eds.). California Riparian Systems. Univ. Calif. Press, Berkeley, CA.

AZ11
Ohmart, R.D., B.W. Anderson, and W.C. Hunter. 1985. Influence of agriculture on waterbird, wader, and shorebird use along the lower Colorado River. pp. 117-122 In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. I B

AZ18
Piest, L.A. and L.K. Sowls. 1985. Breeding duck use of a sewage marsh in Arizona. J. Wildl. Manage. 49:580-585.

AZ21
Hunter, W.C., B.W. Anderson, and R.D. Ohmart. 1985. Summer avian community composition of Tamarix habitats in three southwestern desert riparian systems. pp. 128-134 In: R.R. Johnson, C.D.Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. B

AZ22
Warren, P.L. and L.S. Anderson. 1985. Gradient analysis of a Sonoran desert wash. pp. 150-155. In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. PW

ARIZONA (continued)

AZ23
Johnson, R.R. and L.T. Haight. 1985. Avian use of xeroriparian ecosystems in the North American warm deserts. p. 156-160 In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. B

AZ25
Stevens, L.E. and G.L. Waring. 1985. The effects of prolonged flooding in the riparian plant community in Grand Canyon. pp. 81-86 In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre. Riparian Ecosystems and their Management: Reconciling Conflicting Uses. Gen. Tech Rep. RM-120, USDA Forest Serv., Fort Collins, CO. I PW

AZ25
Stevens, L.E. 1989. Mechanisms of riparian plant community organization and succession in the Grand Canyon, Arizona. Ph.D. Diss., Northern Arizona Univ., Flagstaff. 124 pp.

AZ25
Warren, P.L. and L.S. Anderson. 1985. Gradient analysis of a Sonoran desert wash. pp. 150-155. In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. PW

AZ26-29
Hunter, W.C. 1988. Dynamics of bird species assemblages along a climactic gradient: A Grinnellian niche approach. M.S. Thesis, Arizona State Univ., 103 pp. B

AZ30
Hunter, W.C., B.W. Anderson, and R.D. Ohmart. 1987. Avian community structure changes in a mature floodplain forest after extensive flooding. J. Wildl. Manage. 51(2):495-502. B

AZ31-32
Wilhelm, M., S.R. Lawry, and D.D. Hardy. 1988. Creation and management of wetlands using municipal wastewater in northern Arizona: A status report. pp. 154-159. In: J. Zelazny and J.S. Feierabend (eds.). Increasing Our Wetland Resources. National Wildlife Federation, Washington, D.C.

AZ33-34
Carothers, S.W., A.M. Phillips, III, B.G. Phillips, R.A. Johnson, C.S. Babcock, and M.M. Sharp. 1982. Riparian ecology of the San Fransico River, Frisco Hot Springs, New Mexico to the Martinez Ranch, Arizona.

AZ33-34
James Montgomery Consulting Engineers. 1985. Wildlife and fishery studies, Upper Gila water supply project. Part 2: Fisheries. Unpub., SWCA, Inc. and James Montgomery Engr., Boulder City, NV.
AZ33-34
Johnson, T.B. (ed.). 1981. Final report for the biological survey of the George Whittell Wildlife Preserve. Graham and Pinal Counties, AZ. Map SWCA-2.

AZ33-34
Mills, G.S., S. Sutherland, and R.B. Spicer. 1985. Wildlife and fishery studies, Upper Gila water supply project. Part 1: Terrestrial wildlife. Unpub., SWCA, Inc. and James Montgomery Engr., Boulder City, NV. Map SWCA-1.

AZBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

AZBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C.
AZBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

AZBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

ARIZONA (continued)

AZCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Anderson, B.W., A. Higgins, and R.D. Ohmart. 1977. Avian use of salt cedar communities in the lower Colorado River valley. pp. 128-136 In: R.R. Johnson and D.A. Jones (Tech. Coords.). Importance, Preservation, and Management of Riparian Habitat: A Symposium. Gen. Tech. Rep. RM-43. USDA For. Serv., Fort Collins, CO. B

Anderson, B.W. and R.D. Ohmart. 1977. Vegetation structure and bird use in the lower Colorado River valley. pp. 23-34 In: R.R. Johnson and D.A. Jones (Tech. Coords.). Importance, Preservation, and Management of Riparian Habitat: A Symposium. Gen. Tech. Rep. RM-43. USDA For. Serv., Fort Collins, CO. B

Anderson, B.W. and R.D. Ohmart. 1985. Managing riparian vegetation and wildlife along the Colorado River: synthesis of data, predictive models, and management. pp. 123-127 In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO.

Bloss, D.A. and J.D. Brotherson. 1979. Vegetation response to a moisture gradient on an ephemeral stream in central Arizona. Great Basin Nat. 39:161-176.

Brady, W., D.R. Patton, and J. Paxson. 1985. The Development of Southwestern Riparian Gallery Forests. pp. 39-43 In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. PW

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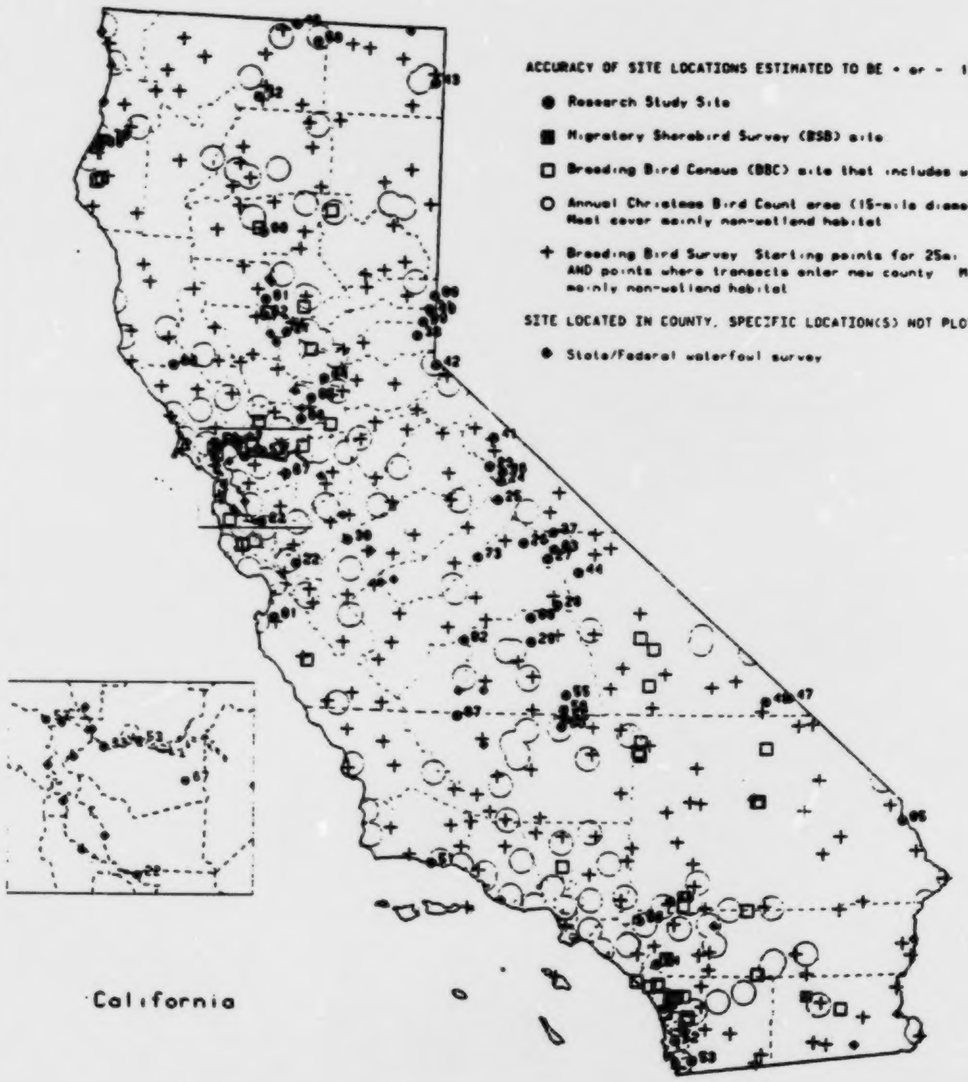
Inland Wetlands Having Biological Community Measurements



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

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- ○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat
- + Breeding Bird Survey Starting points for 25mi transects AND points where transects enter new county Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

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Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

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CALIFORNIA

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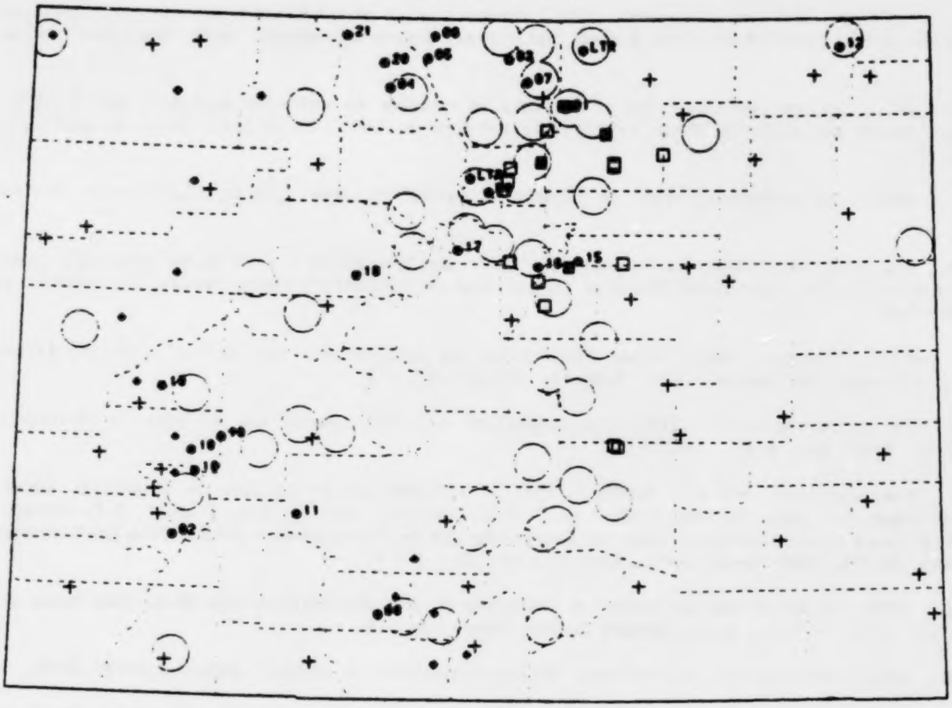
Inland Wetlands Having Biological Community Measurements

Colorado



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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria   Cartography Jeff Irish   July 1988

COLORADO

Mapped

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COBSB1-
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COBW1-
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COLTR
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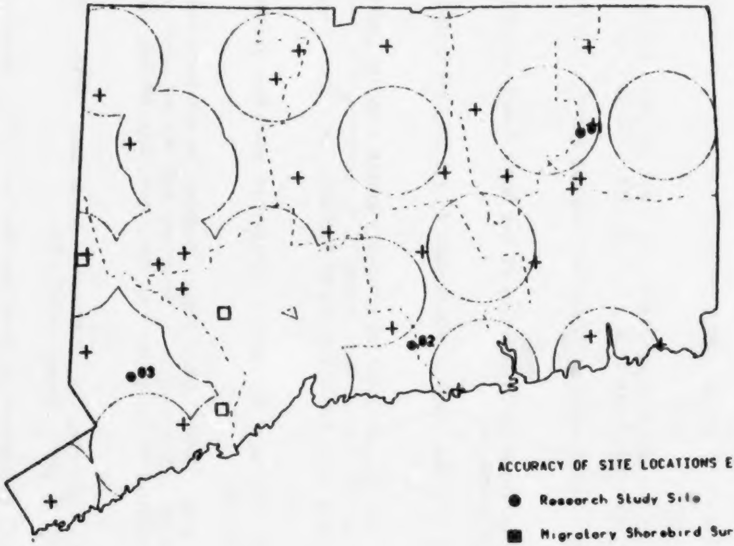
Inland Wetlands Having Biological Community Measurements

Connecticut



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

● Research Study Site

■ Migratory Shorebird Survey (BSB) site

□ Breeding Bird Census (BBC) site that includes wetland

○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat

+ Breeding Bird Survey Starting points for 25mi transects AND points where transects enter new county Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

◆ State/Federal waterfowl survey

This map does NOT portray ALL wetland sampling sites Emphasis is on sites where community-level data were collected See text for inclusion criteria

Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

CONNECTICUT

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CT1
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CT3
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CTBBS1-
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CTBW1-
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CTCBC1-
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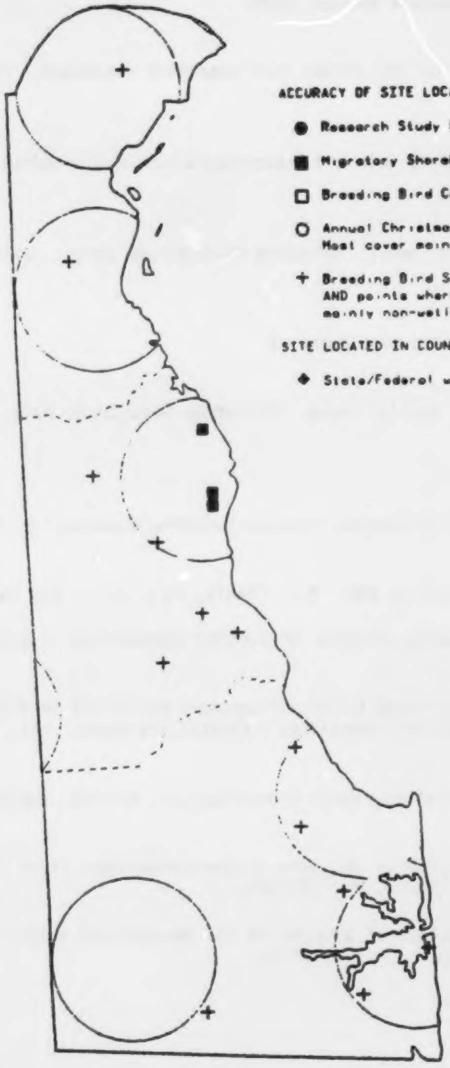
Inland Wetlands Having Biological Community Measurements



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Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

DELAWARE

Mapped

DEBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

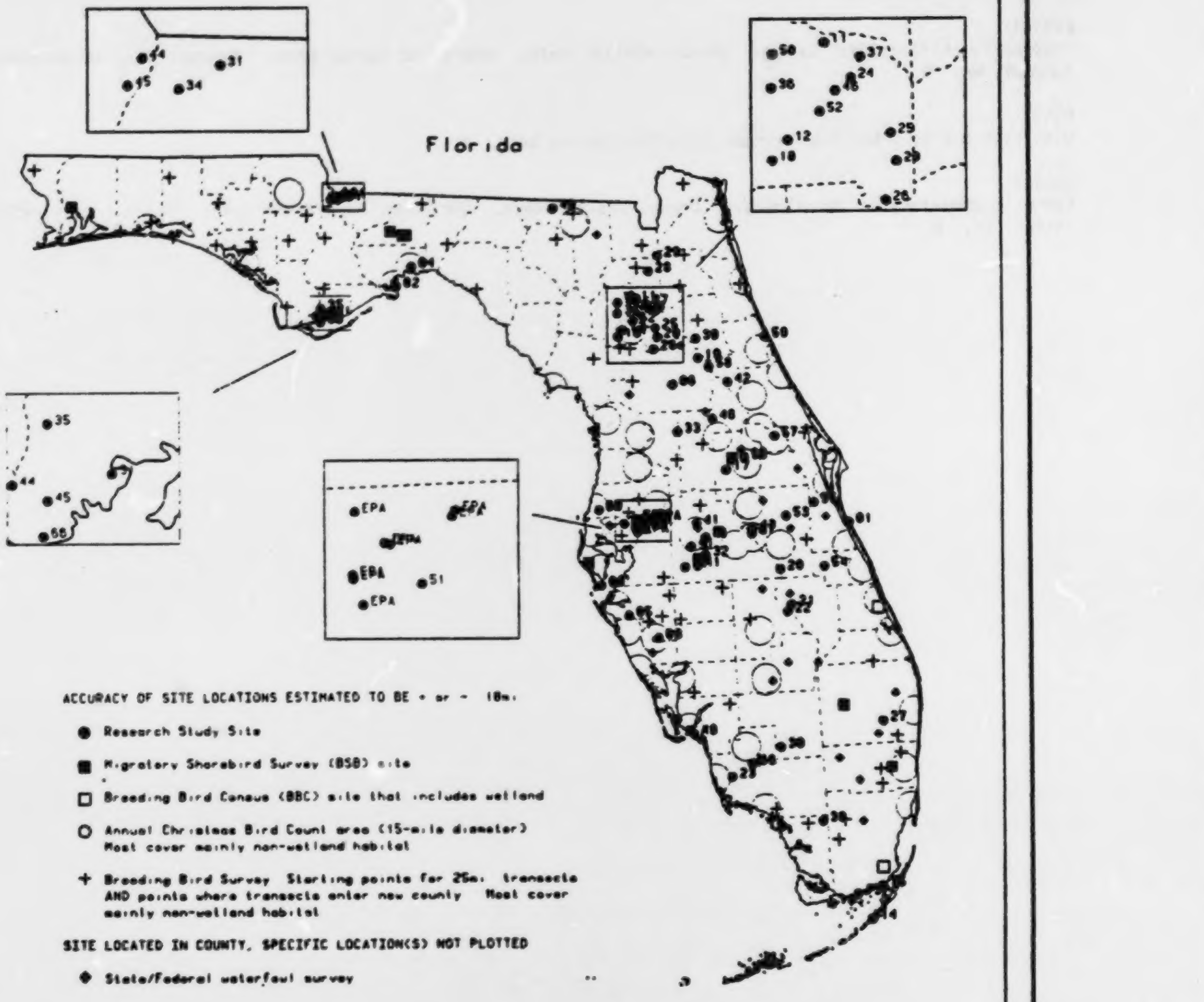
DEBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

DEBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

DEBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

DECBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Inland Wetlands Having Biological Community Measurements



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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

FLORIDA

Mapped

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FL17-20
Canfield, D.E., K.A. Langeland, M.J. Maccina, W.T. Haller, and J.V. Shireman. 1983. Trophic state classification of lakes with aquatic macrophytes. Can. J. Fish. Aquat. Sci. 40(10):1713-1718. PM I

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FLBBC1-

Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

FLORIDA (continued)

FLBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

FLBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

FLBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

FLCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

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Not Mapped

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Dalrymple, G.H. 1988. The herpetofauna of Long Pine Key, Everglades National Park, in relation to vegetation and hydrology. pp. 72-86 In: R.C. Szaro, K. E. Severson, D.R. Patton (tech. coords.). Management of Amphibians, Reptiles, and Small Mammals in North America. Gen. Tech. Rep. RM-166, USDA Forest Serv., Fort Collins, CO.

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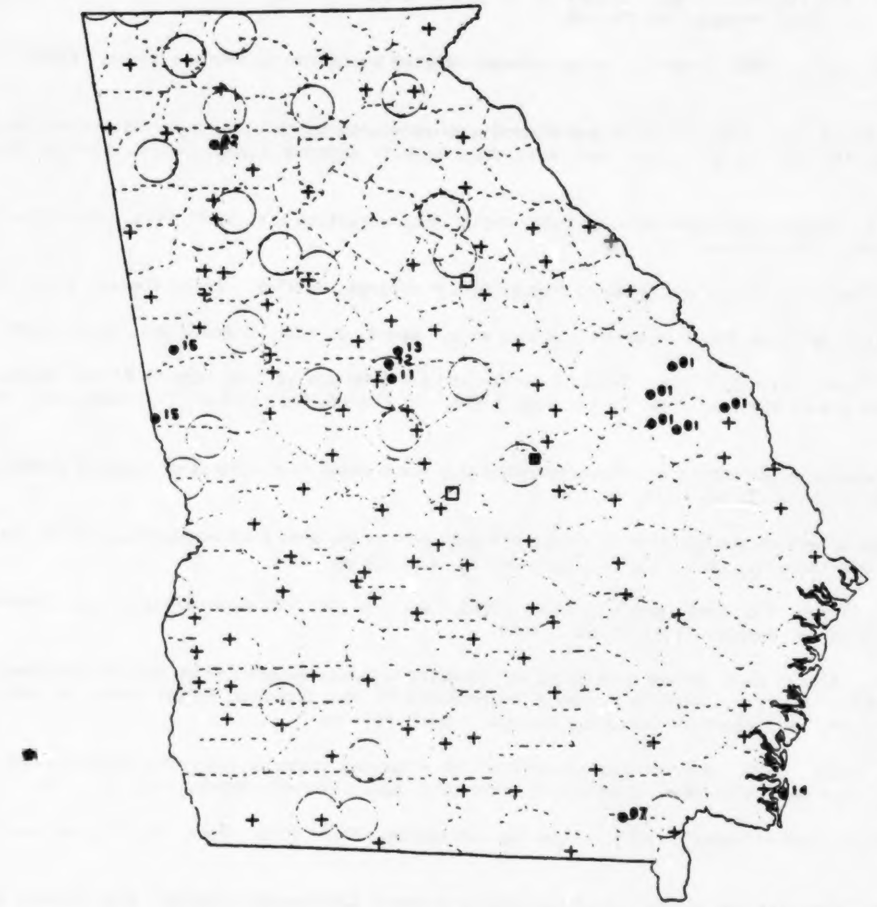
Inland Wetlands Having Biological Community Measurements



Georgia

ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi.

● Research Study Site

■ Migratory Shorebird Survey (BSB) site

☐ Breeding Bird Census (BBC) site that includes wetland

○ Annual Christmas Bird Count area (15-mile diameter)
Most cover mainly non-wetland habitat

+ Breeding Bird Survey Starting points for 25mi. transects
AND points where transects enter new county Most cover
mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

◆ State/Federal waterfowl survey

This map does NOT portray ALL wetland sampling sites
Emphasis is on sites where community-level data were
collected See chapter 1 for inclusion criteria

Sites are referenced by code number to the accompanying
state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

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GABBS1-
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GABSB1-
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GEORGIA (continued)

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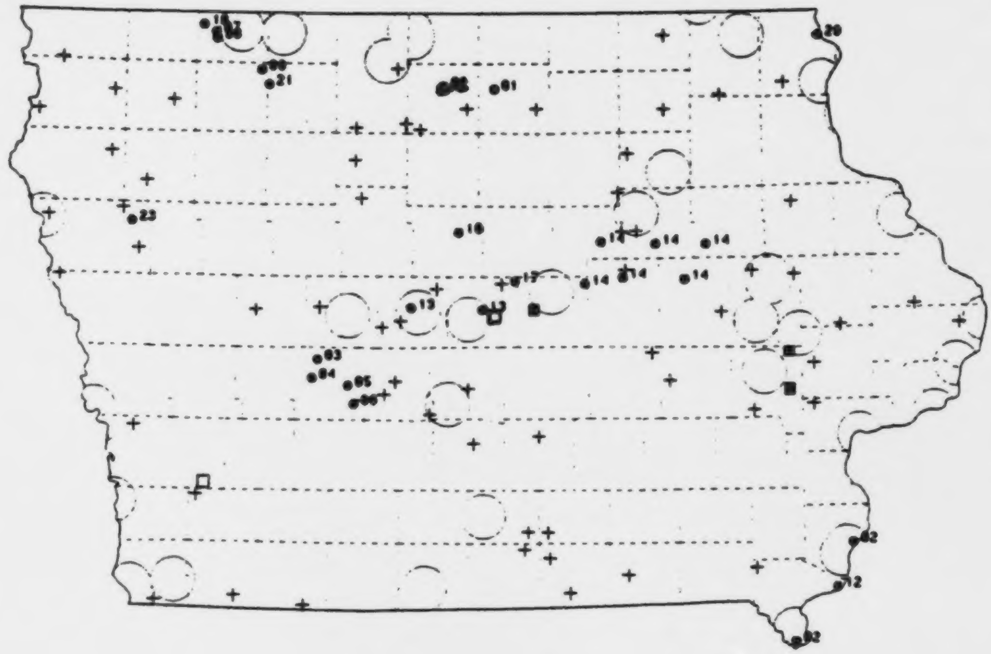
Inland Wetlands Having Biological Community Measurements

Iowa



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Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

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IOWA

Mapped

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IABBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

IABSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

IABW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

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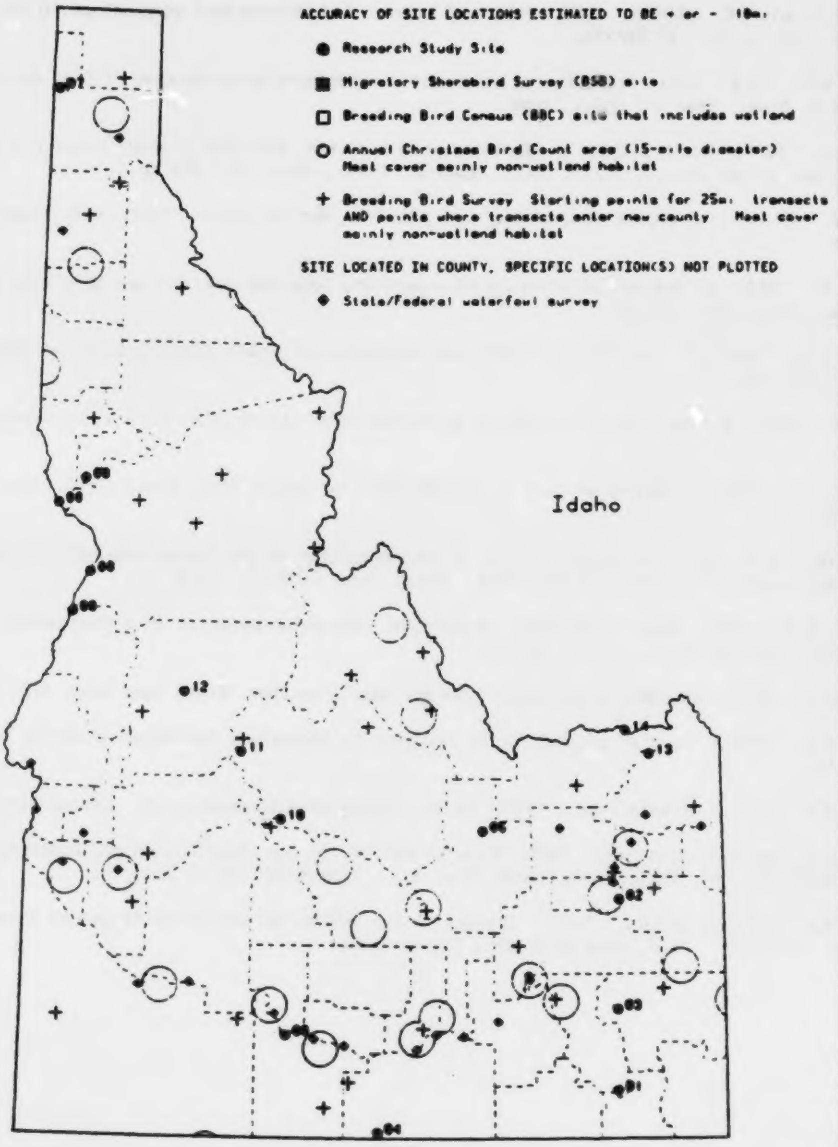
Inland Wetlands Having Biological Community Measurements


ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi
● Research Study Site
■ Migratory Shorebird Survey (MSS) site
☐ Breeding Bird Census (BBC) site that includes wetland
○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat
+ Breeding Bird Survey Starting points for 25mi transects AND points where transects enter new county Most cover mainly non-wetland habitat
SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED
◆ State/Federal waterfowl survey
Idaho
01
12
11
10
14
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02
03
01
04

This map does NOT portray ALL wetland sampling sites Emphasis is on sites where community-level data were collected See chapter 1 for inclusion criteria

Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Rontoria Cartography Jeff Irish July 1988

IDAHO

Mapped

ID1
Wolf, K. 1955. Some effects of fluctuating and falling water levels on waterfowl production. J. Wildl. Manage. 19(1):13-23. B I

ID2
Halford, D.K., O.D. Markham, and R.L. Dickson. 1982. Radiation doses to waterfowl using a liquid radioactive waste disposal area. J. Wildl. Manage. 46(4):905-913. B

ID3
Gregg, W.W. and F.L. Rose. 1982. The effects of aquatic macrophytes on the stream microenvironment. Aquat. Bot. 14:309-324. PM

ID4-5
Jensen, S., R. Ryel and W.S. Platts. 1989. Classification of riverine/riparian habitat and assessment of nonpoint source impacts, North Fork Humboldt River, Nevada. USDA Forest Service, Intermountain Res. Stn., Boise Fisheries Unit.

ID6
Army Corps. of Engineers. 1983. Interim Feasibility Report. Clear Lakes Hydropower, Snake River, ID.

ID7
Rumely, J.H. 1956. Plant ecology of a bog in northern Idaho. Ph.D. Dissertation, Washington St. Univ., Pullman. 85 pp.

ID8
Miller, T.B. 1976. Ecology of riparian communities dominated by white alder in western Idaho. M.S. Thesis, Univ. Idaho, Moscow. 154 pp.

ID9
Manuel-Faler, C.Y. 1981. Production and fate of aquatic macrophytes in Deep Creek, Idaho. Ph.D. Dissertation, Idaho St. Univ., Pocatello. 127 pp.

ID10
Tuhy, J.S. 1981. Stream bottom community classification for the Sawtooth Valley, Idaho. M.S. Thesis, Univ. Idaho, Moscow. 175 pp.

ID11
Tuhy, J.S. 1982. Riparian Classification for the Upper Salmon/Middle Fork Salmon River Drainages, Idaho. Smithfield Associates, Smithfield, UT. 153 pp.

ID12-13
Mutz, K. and J. Queiroz. 1983. Riparian Community Classification for the Centennial Mountains and South Fork Salmon River, Idaho. Meiji Resource Consultants, Layton, UT. 96 pp.

ID14
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IDBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

IDBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

IDCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

IDAHO (continued)

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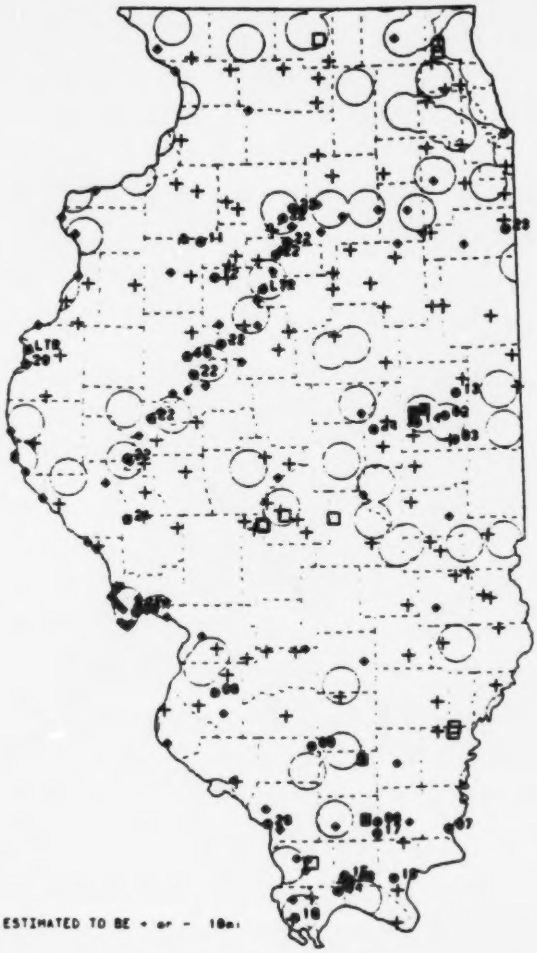
Inland Wetlands Having Biological Community Measurements



Illinois

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Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

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ILLINOIS

Mapped

IL2
Clark, W.D. and J.R. Karr. 1979. Effects of highways on Red-Winged Blackbirds and Horned Lark populations. Wilson Bull. 91(1):143-145. B I

IL3
Wiley, M.J., R.W. Gorden. S.W. Waite, and T. Powers. 1984. The relationship between aquatic macrophytes and sport fish production in Illinois ponds: A simple model. N. Amer. J. Fish. Manage. 4:111-119. F

IL4
Brown, S. and L. Giese. 1988. Tree Growth Rates and Regeneration of Buttonland Swamp, Southern Illinois. Final Report to Illinois Dept. of Conserv., Cache River Basin Study. Dept. of Forestry, Univ. of Illinois, Urbana, IL. 67 pp. PW

IL5
Yeager, L.E. 1949. Effect of Permanent Flooding on a River Bottom Timber Area. Bull. Ill. Nat. History Surv., Urbana, IL 25(2):33-65. PW

IL6-8
Jones, D.W., M.J. McElligott, and R.H. Mannz. 1985. Summary of Biological, Chemical, and Morphological Characterizations of 33 Surface-Mine Lakes in Illinois and Missouri. pp. 211-238 In: R.P. Brooks, D.E. Samuel, and J.B. Hill (eds.). Wetlands and Water Management on Mined Lands. Penn. St. Univ., University Park, PA. R

IL9-12
Lawrence, J.S., W.D. Kilmstra, W.G. O'Leary, and G.A. Perkins. 1985. Contribution of surface-mined wetlands to selected avifauna in Illinois. pp. 317-326 Penn. St. Univ., University Park, PA. In: R.P. Brooks, D.E. Samuel, and J.B. Hill (eds.). Wetlands and Water Management on Mined Lands. Penn. St. Univ., University Park, PA. B

IL13-14
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IL15-16
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IL17
Urbanek, R.P. and W.D. Klimstra. 1986. Vertebrates and vegetation on a surface-mined area in southern Illinois. Trans. Illinois Acad. Sci. 79(3):175-187. B P

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IL19
Pinkowski, R.H., G.L. Rolfe, and L.E. Arnold. 1985. Effect of feedlot runoff on a southern Illinois forested watershed. J. Environ. Qual. 14(1):47-54. I P

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Grubaugh, J.W., R.V. Anderson, D.M. Day, K.S. Lubinski, and R.E. Sparks. 1986. Production and fate of organic material from Sagittaria latifolia and Nelumbo lutea on Pool 19, Mississippi River. J. Freshw. Ecol. 3(4):477-484. SO P

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ILLINOIS (continued)

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IL23
Kwak, Thomas J. 1988. Lateral movement and use of floodplain habitat by fishes of the Kankakee River, Illinois. Amer. Midl. Nat. 120(2):241-149.

IL24
Uetz, G.W., K.L. Van der Laan, G.F. Summers, P.A. Gibson, and L.L. Getz. 1979. The effects of flooding on floodplain arthropod distribution, abundance and community structure. Amer. Midl. Nat. 101(2):286-299. AI

IL25
Greenfield, D.W. and J.D. Rogner. 1984. An assessment of the fish fauna of Lake Calumet and its adjacent wetlands, Chicago, IL: Past, Present and Future. Trans. Illinois Acad. Sci. 77(1):77-93. F TS

IL26
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IL27
Cole, C.A. 1988. Wetland ecosystem development on a reclaimed surface coal mine in southern Illinois.Ph.D. Diss., Dept. of Zoology, Southern Illinois Univ., Carbondale, IL. 284 pp.

ILBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

ILBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

ILBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

ILBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

ILCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

ILLTR
Anderson, R.V. et al. In Process. Long Term Environmental Research Wetland Site: Illinois (Pool 19) LTER Site. Dept. of Biol. Sci., Western Illinois Univ., Macomb, IL. AI F B P

ILLTR
Sparkes, R.E. et al. In Process. Long Term Environmental Research Wetland Site: Illinois and Mississippi Rivers LTER Site. Illinois Nat. Hist. Surv., Macomb, IL. MI AI F B P

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Angermeier, P.L. 1988. Spatiotemporal variation in habitat selection by fishes in small Illinois streams. pp. 52-60 In: W.J. Matthews and D.C. Heins (eds.). Community and Evolutionary Ecology of North American Stream Fishes. Univ. Oklahoma Press, Norman.

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ILLINOIS (continued)

O'Leary, W.G. 1984. Wuterfowl habitats provided by surface mine wetlands in southwestern Illinois. MA Thesis, Dept of Zoology, Southern Illinois Univ., Carbondale, IL. 141 pp. B P

Osborne, L.L., and E.E. Herricks. 1983. Streamflow and Velocity as Determinants of Aquatic Insect Distribution and Benthic Community Structure in Illinois. Water Resources Center, Univ. Ill., Urbana-Champaign. 230pp.

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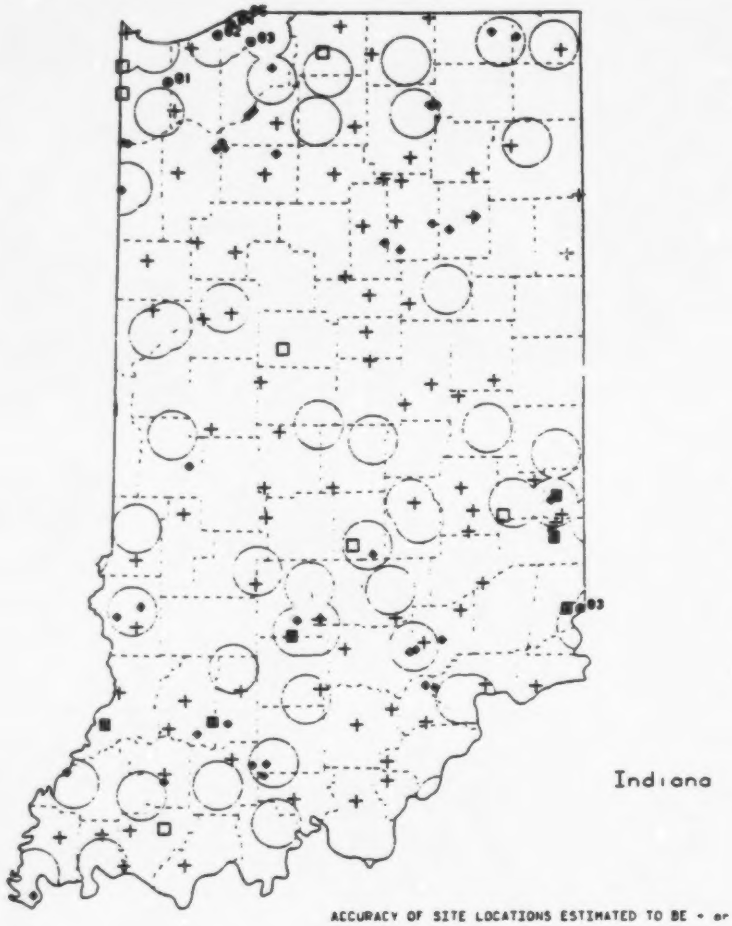
Inland Wetlands Having Biological Community Measurements



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INDIANA

Mapped

IN1
Wilcox, D.A., N.B. Pavlovic, and M.L. Mueggler. 1985. Selected ecological characteristics of *Scirpus cyperinus* and its role as an invader of disturbed wetlands. Wetlands 5:87-97. PE

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IN2
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IN2
R.D. Hiebert, D.A. Wilcox, and N.B. Pavlovic. 1986. Vegetation patterns in and among pannes (Calcareous Intradunal Ponds) at the Indiana Dunes National Lakeshore, Indiana. Amer. Midl. Nat. 116(2):276-281.

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Jackson, S.T., R.P. Futyma, and D.A. Wilcox. 1988. A paleoecological test of a classical hydrosere in the Lake Michigan Dunes. Ecol. 69(4):928-936. P

IN2,5
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IN3
Wilcox, D.A. 1986. The effects of deicing salts on vegetation in Pinhook Bog, Indiana. Can. J. Bot. 64:865-874. P I

IN4
Wilcox, D.A. and R.E. Andrus. 1987. The role of *Sphagnum fimbriatum* of secondary succession in a road salt impacted bog. Can. J. Bot. 65:2270-2275. PB I

INBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

INBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

INBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

INBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

INCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Clifford, H.F. 1966. The ecology of invertebrates in an intermittent stream. Invest. Indiana Lakes & Streams 7:57-98. AI

INDIANA (continued)

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Hendrickson, W.H. and D.A. Wilcox. 1979. Relationship between some physical properties and the vegetation found in Cowles Bog National Natural Landmark, Indiana. Proc. Sec. Conf. Sci. Res. Nat. Parks. 5:642-666. P

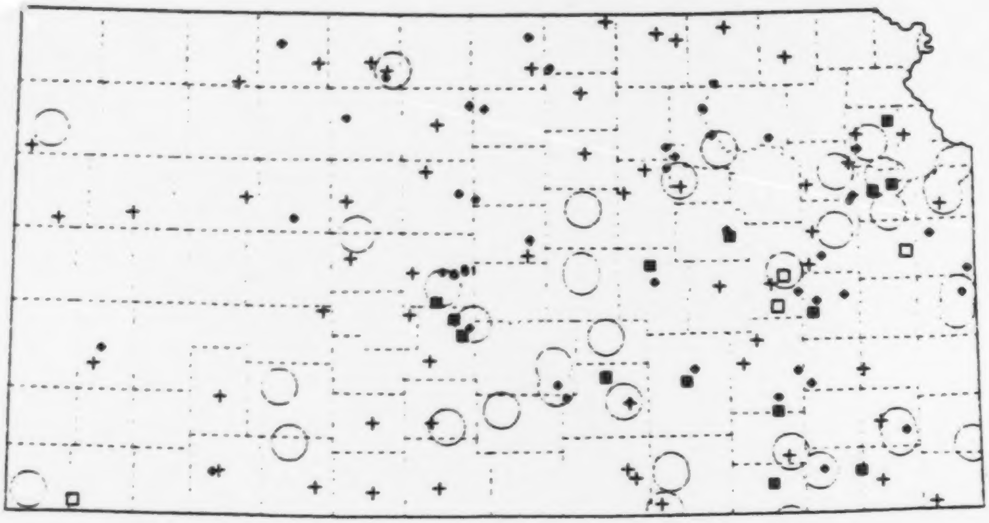
Hiebert, R.D., D.A. Wilcox, and N.B. Pavlovic. 1986. Vegetation patterns in and among pannes (calcareous intradunal ponds) at the Indiana Dunes National Lakeshore. Amer. Nat. 116:276-281. PE

Landers, D.H. 1982. Effects of naturally senescing aquatic macrophytes on nutrient chemistry and chlorophyll a of surrounding waters. Limnol. Oceanogr. 27:428-439. PM

Lindsey, A.A., R.O. Petty, D.K. Sterling, and W. Van Asdall. 1961. Vegetation and environment along the Wabash and Tippecanoe rivers. Ecol. Monogr. 31:105-156. PW

Inland Wetlands Having Biological Community Measurements

Kansas



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KANSAS

Mapped

KS1
Kansas Biological Survey and Kansas Geological Survey. 1987. Cheyenne Bottoms: An Environmental Assessment. Univ. of Kansas, Rep. # 32. Submitted to the KS Fish and Game Comm.

KS2-15
Shipley, F.S. 1980. Habitat distribution and fecundity in a marsh-upland bird species series. Ph.D. Diss., Kansas State Univ. 140 pp. B

KSBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

KSBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

KSBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

KSBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

KSCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Anderson, R.G. 1954. A palynological study of a fresh water marsh in Atchison County, Kansas. M.A. Thesis, Dept. of Bot., Univ. Kansas, 21 pp.

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Inland Wetlands Having Biological Community Measurements

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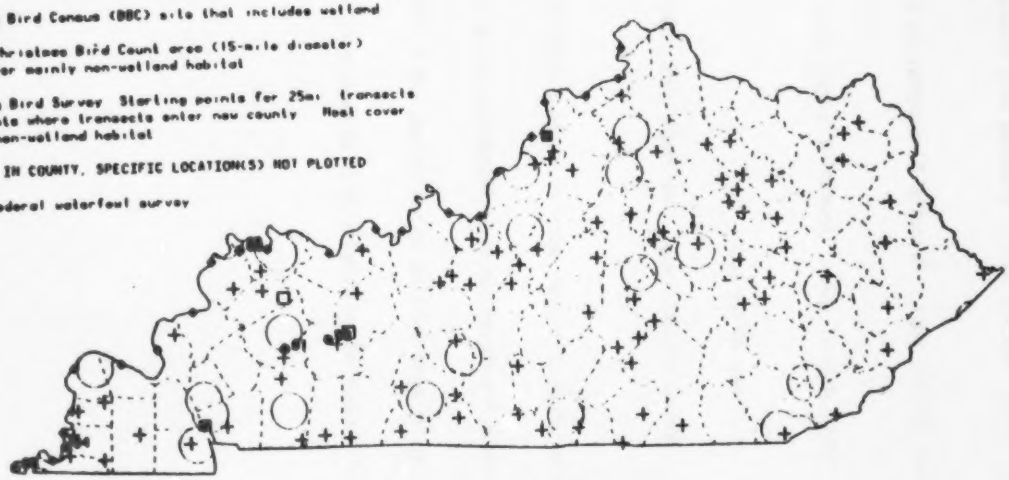

Kentucky

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Emphasis is on sites where community-level data were
collected See chapter 1 for inclusion criteria

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state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

July 1988

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish

KENTUCKY

Mapped

KY1-3
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KYBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

KYBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

KYCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

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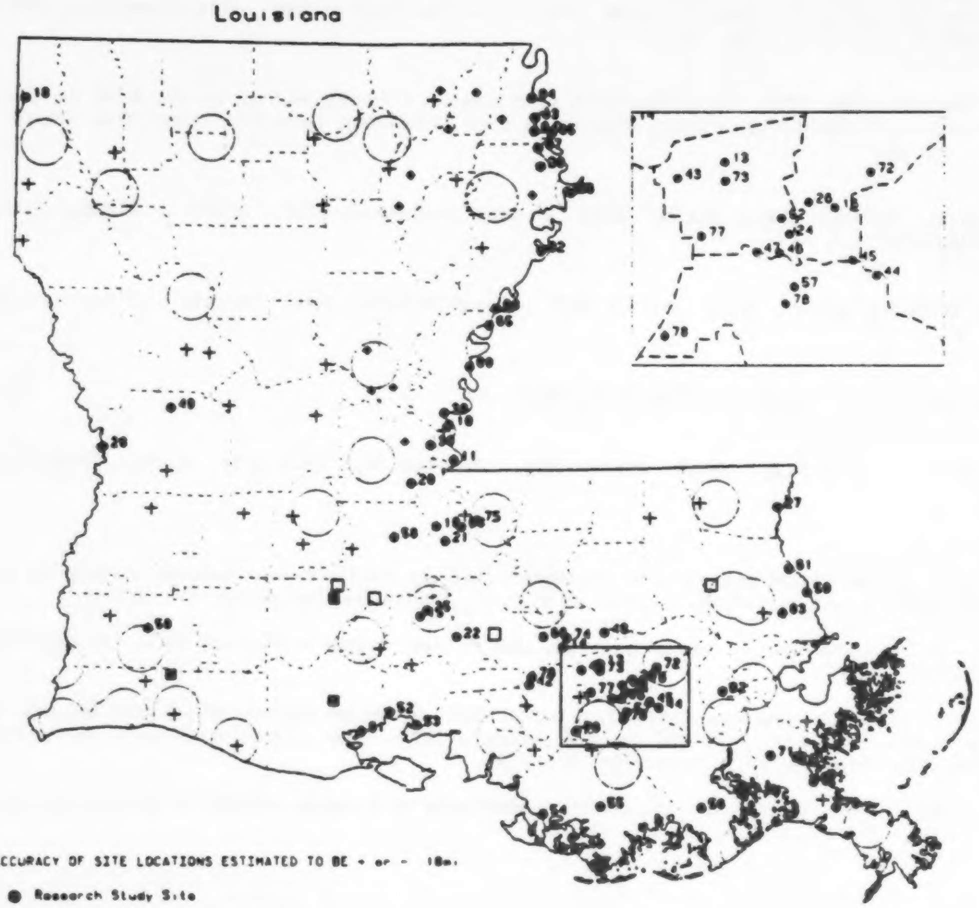
Inland Wetlands Having Biological Community Measurements



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 18m.

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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

LOUISIANA

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LABBS1-
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LABSB1-
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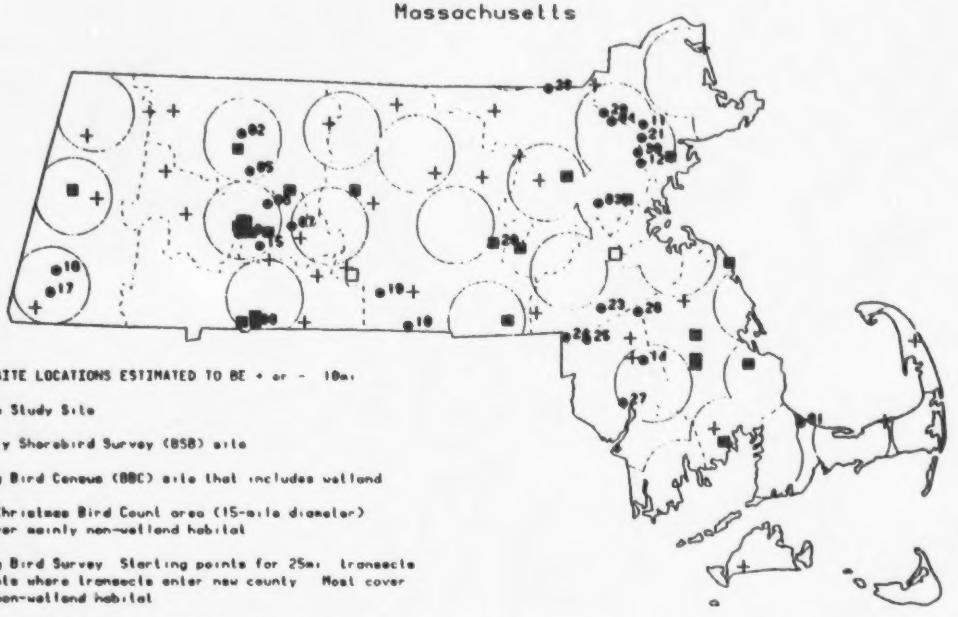
Inland Wetlands Having Biological Community Measurements



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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

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MASSACHUSETTS

Mapped

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MABBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

MASSACHUSETTS (continued)

MABSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

MABW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

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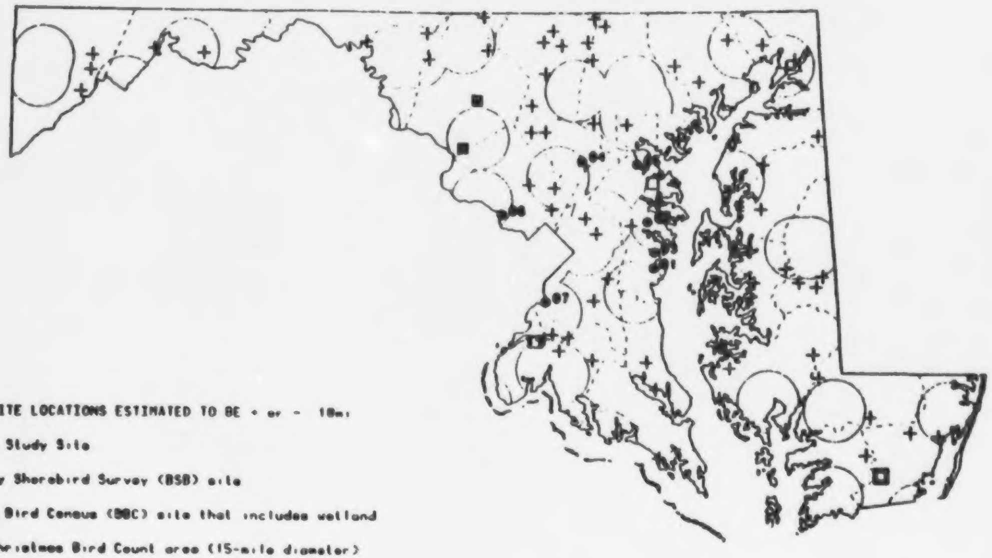
Inland Wetlands Having Biological Community Measurements

Maryland



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

- ● Research Study Site
- ■ Migratory Shorebird Survey (MSB) site
- ☐ Breeding Bird Census (BBC) site that includes wetland
- ○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat
- + Breeding Bird Survey Starting points for 25mi transects AND points where transects enter new county Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

- ◆ State/Federal waterfowl survey

This map does NOT portray ALL wetland sampling sites Emphasis is on sites where community-level data were collected See text for inclusion criteria

Sites are referenced by code number to the accompanying state bibliography

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MARYLAND

Mapped

MD1-2
Whigham, D.F. and C.J. Richardson. 1988. Soil and plant chemistry of an Atlantic white cedar wetland on the Inner Coastal Plain of Maryland. Can. J. Bot. 66:568-576. P

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Southwick, C.H. and F.W. Pine. 1975. Abundance of submerged vascular vegetation in the Rhode River from 1966 to 1973. Chesapeake Sci. 16(1):147-151. PM TS

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MD4
Adams, L.W., I.E. Dove, and T.M. Franklin. 1985. Use of urban stormwater control impoundments by wetland birds. Wilson Bull. 97(1):120-122. B

MD5
Bartoldus, C. Unpub. Marley Creek, Maryland Wetland Monitoring, Annadale, VA. P

MDBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

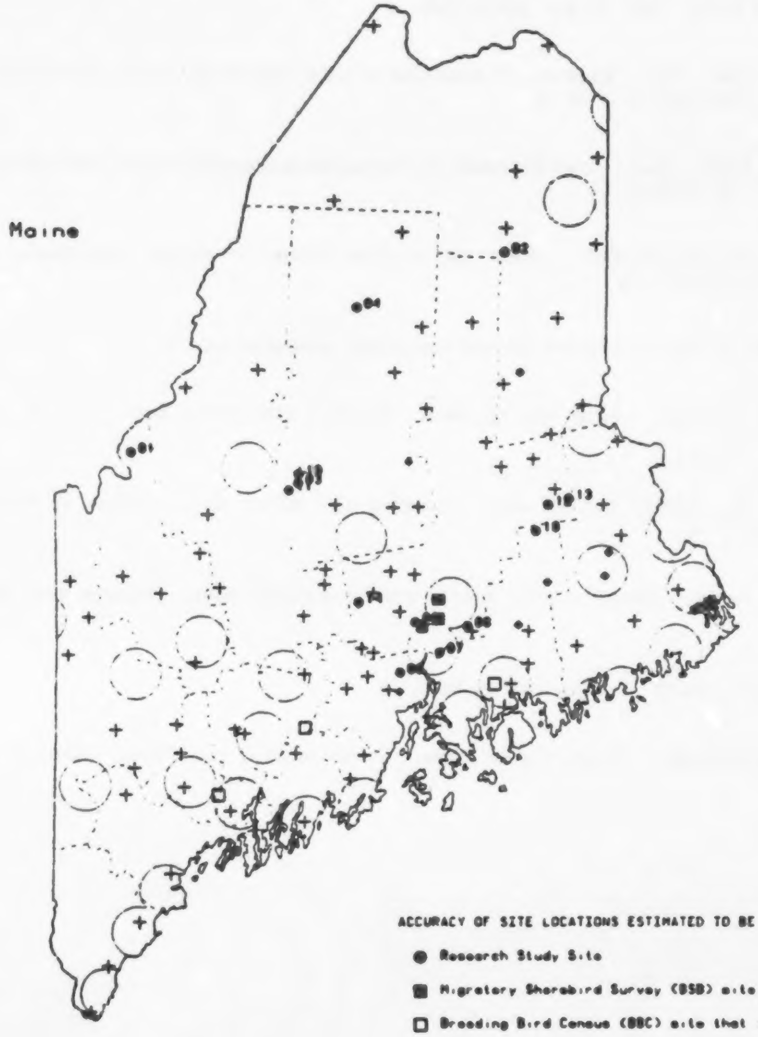
MDBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

MDBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

MDBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

MDCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Inland Wetlands Having Biological Community Measurements



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MAINE

Mapped

ME1
Gibbs, K.E., T.M. Mingo, D.L. Courtemanch, and D.J. Stairs. 1981. The Effects on Pond Macroinvertebrates from Forest Spraying of Carbaryl (Sevin-4-Oil) and its Persistence in Water and Sediment. In: Environ. Monitoring Rep., Marine Forest Serv., Augusta. AI I

ME2-3
Moring, J.R., G.C. Garman, and J. Mullen. 1985. The value of Riparian zones for protecting aquatic systems: general concerns and recent studies in Maine. pp. 315-319 In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre (tech. coords.). Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. F AI

ME4
Hunter, M.L., Jr., J.J. Jones, K.E. Gibbs, and J.R. Moring. 1986. Duckling responses to lake acidification: Do black ducks and fish compete? Oikos 47:26-32. B I

ME4
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Fefer, S.I. 1977. Waterfowl Populations as Related to Habitat Changes in Bog Wetlands of the Moosehorn National Wildlife Refuge. Bull. Life Sci. Agric. Expt. Stn., Univ. Maine, Tech. Bull. 86:16. B

ME6
Ringelman, J.K. and J.R. Longcore. 1982. Movements and wetland selection by brood-rearing Black Ducks. J. Wildl. Manage. 46(3):615-621. B R

ME7-13
Stockwell, S.S. 1985. Distribution and Abundance of Amphibians, Reptiles, and Small Mammals In Eight Types of Maine Peatland Vegetation. M.S. Thesis, Univ. Maine, Oronno, ME. 57 pp. H R

ME14
Jiffry, F. 1984. Loss of freshwater shellfish and some ecological impacts after water drawdown in Lake Sebasticook, Maine. M.S. Thesis, Univ. Maine, Orono. AI I S

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McAuley, D.G. and J.R. Longcore. 1988. Foods of juvenile Ring-Necked Ducks: Relationship to Wetland pH. J. Wildl. Manage. 52(2):177-185. B R

ME15,16
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ME19
Kenlan, K.H., G.L. Jacobson, Jr., and D.F. Brakke. 1983. Aquatic macrophytes and pH as controls of diversity for littoral Cladocerans. pp. 63-83. In: Hendrey. (eds.). Early Biotic Responses to Advancing Lake Acidification. I AI

MEBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

MEBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

MEBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

MAINE (continued)

MEBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

MECBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

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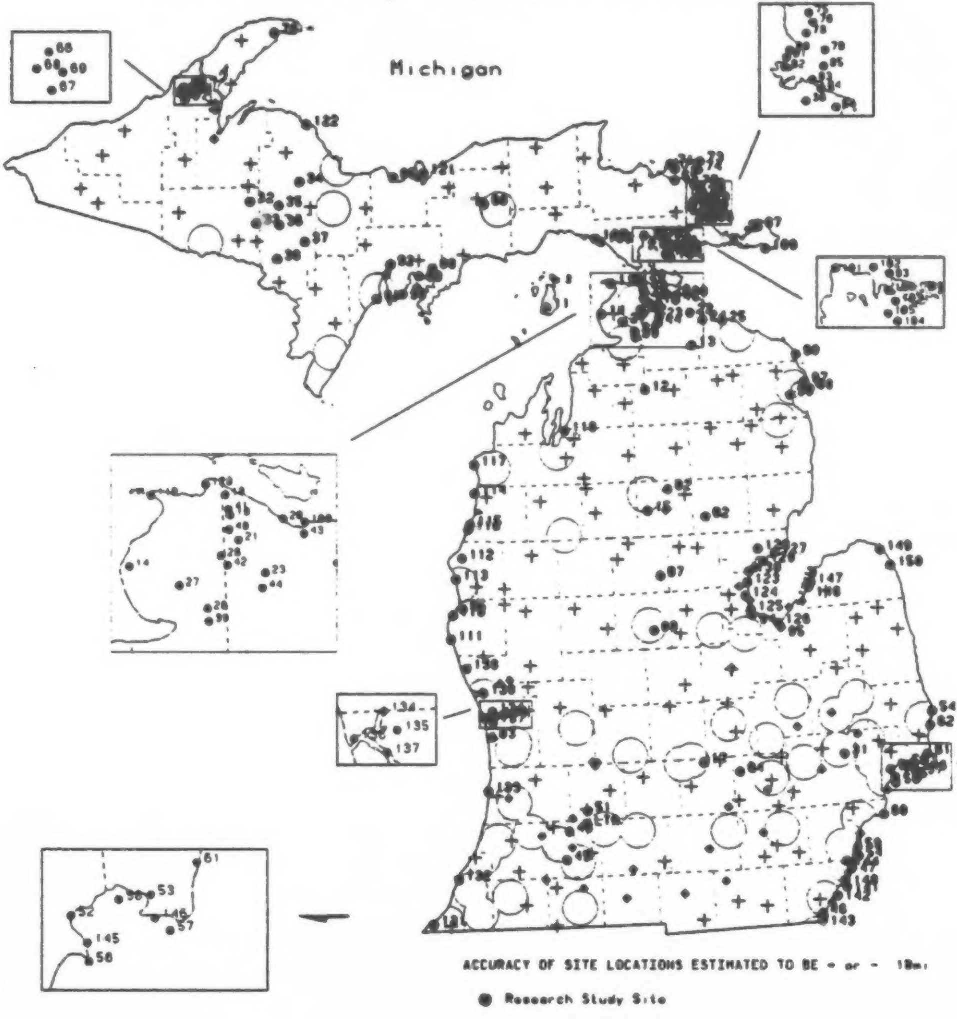
Inland Wetlands Having Biological Community Measurements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ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

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MICHIGAN

Mapped

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Davis, P.B. and C.R. Humphrys. 1977. Ecological effects of highway construction upon Michigan woodlots and wetlands. Michigan State Univ., Agric. Expt. Station, Dept. of Resource Dev. #914. Michigan Agric. Expt. Stn. J. Article #8208. P B I

MI3
Mansfield, P.J. 1984. Reproduction by Lake Michigan fishes in a tributary stream. Trans. Amer. Fish. Soc. 113:231. F

MI4
Burton, T.M. and D.L. King. 1983. Alterations in the biodynamics of the Red Cedar River associated with human impacts during the past 20 years. Ann Arbor Sci. Pub., Ann Arbor, MI. I PM TS

MI5
Chubb, S.L. and C.R. Liston. 1986. Density and distribution of larval fishes in Pentwater Marsh, a coastal wetland on Lake Michigan. J. Great Lakes Res. 12(4):332-343. F

MI6
Beard, E.B. 1963. Duck brood behavior at the Seney National Wildlife Refuge. J. Wildl. Manage. 28:492-497. B

MI7-8
Ewert, D. 1982. Birds in isolated bogs in central Michigan. Amer. Midl. Nat. 108(1):41-50. B

MI9
Richardson, C.J. and B.R. Schwegler. 1986. Algal bioassay and gross productivity experiments using sewage effluent in a Michigan wetland. Water Res. Bull. 22(1):111-120. A T I

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Foran, J.A. and R.H. King. 1982. Regression analysis of the summer population dynamics of *Polyarthra vulgaris* in a northern Michigan bog lake. Hydrobiol. 94:237-246. AI

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Momot, W.T., H. Gowing, and P.D. Jones. 1978. The dynamics of crayfish and their role in ecosystems. Amer. Midl. Nat. 99(1):10-35. AI

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Schwintzer, C.R. and G. Williams. 1974. Vegetation changes in a small bog from 1917 to 1972. Amer. Midl. Nat. 12:447-459. P

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Croson, S.C. 1975. Distribution and abundance of insects in a wetland ecosystem. M.S. Thesis, Univ. Michigan, Ann Arbor, MI, 142 pp.

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MI15
Richardson, C.J., J.A. Kadlec, W.A. Wentz, J.P.M. Chamie, and R.H. Kadlec. 1976. Background ecology and the effects of nutrient additions on a central Michigan wetland. pp. 34-72. In: LeFor, M.W., W.C. Kennard and T.B. Helfgott (eds.). Proc. Third Wetland Conf., Inst. Water Res., Univ. of Connecticut, Storrs, CT. Rep. No. 26.

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Kadlec, R. H., D. L. Tilton, and B. R. Schwegler. 1979. Three-year summary of pilot scale operations at Houghton Lake. Report Nat. Sci. Foundation. NTIS PB295965.

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Rabe, D.L. 1989. Impact of wastewater discharge upon a northern Michigan wetland wildlife community. Report Michigan DNR, Lansing. 20 pp.

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Rosman, L. 1978. Impact assessment of a northern Michigan wetland invertebrate and vertebrate fauna receiving secondarily treated sewage effluent. pp. 38-85. In: Kadlec, R.H. et. al (eds.). First Ann. Oper. Rep., Houghton Lake Wetland Treatment Proj.

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MI15
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MI18-28
Henebry, M.S., J. Cairns, Jr., C.R. Schwintzer, and W.H. Yongue, Jr. 1981. A comparison of vascular vegetation and protozoan communities in some freshwater wetlands of northern lower Michigan. Hydrobiol. 83:353-375. MI P

MI30
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MI31
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MI32-38
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Henebry, M.S., J. Cairns, Jr., C.R. Schwintzer, and W.H. Yongue, Jr. 1981. A comparison of vascular vegetation and protozoan communities in some freshwater wetlands of northern lower Michigan. Hydrobiol. 83:353-375. MI P

MI39-44
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MI46
King, D.R. and G.S. Hunt. 1967. Effect of Carp on vegetation in a Lake Erie marsh. J. Wildl. Manage. 31:181-188. F

MI47
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MI48
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MI51
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MI51
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MI51
Beer, S. and R.G. Wetzel. 1982. Photosynthesis in submersed macrophytes of a temperate lake. Plant Physiol. 70:488-492.

MI52-53
Poe, T.P., C.O. Hatcher, C.L. Brown, and D.W. Schloesser. 1986. Comparison of species composition and richness of fish assemblages in altered and unaltered littoral habitats. J. Freshw. Ecol. 3(4):525-536. F I

MI54
Schloesser, D.W. and B.A. Manny. 1986. Distribution of submersed macrophytes in the St. Clair-Detroit River System. J. Freshw. Ecol. 3(4):537-543. PM

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MI54
Schloesser, D.W. and B.A. Manny. 1989. Potential effects of shipping on submersed macrophytes in the St. Clair and Detroit rivers of the Great Lakes. Contrib. 696 Nat. Fish. Res. Center - Great Lakes, Ann Arbor, Michigan, Acad., Sci., Arts, Lett. 0026-2005/89, Michigan Academician XXI, pp. 101-108. PM I

MI55
Kadlec, R.H. and F.B. Bevis. 1990. Wetlands and wastewater: Kinross, Michigan. Wetlands 10:77-92.

MI56-58
Brown, C.L., T.P. Poe, J.R.P. French III, and D.W. Schloesser. 1988. Relationships of phytomacrofauna to surface area in naturally occurring macrophyte stands. J. North Amer. Benthol. Soc. 7(2):129-139. AI

MI59-62
Nichols, S.J., D.W. Schloesser, and P.L. Hudson. 1989. Submersed macrophyte communities before and after an episodic ice jam in the St. Clair and Detroit rivers. Nat. Fish. Res. Center - Great Lakes, U.S. Fish & Wildl. Serv., Ann Arbor, MI. PM

MI63
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MI64
Robertson, G.P. et al. In Process.

MI46, 47, 53, 56, 65-150
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MIBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

MIBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

MIBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

MIBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

MICBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

MILTR
Robertson, G.P., et al. In Process. Long Term Environmental Research Wetland Site: Kellogg Biological Station LTER Site. Michigan State Univ., Hickory Corners, MI. MI AI F A

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Curtis, R. 1963. Waterfowl productivity and a comparison of waterfowl habitats at Rose Lake Wildlife Experiment Station. Michigan Dept. Conserv. Game Div., Rep. No. 2398, Lansing, MI. 85 pp. B P

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Guntenspergen, G.R., W. Kappel, and F. Stearns. 1980. Response of a bog to application of lagoon sewage: The Drummond Project--an operational trial. pp. 559-561 In: Proceedings of the 6th Internat. Peat Congress, Duluth, MN.

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MICHIGAN (continued)

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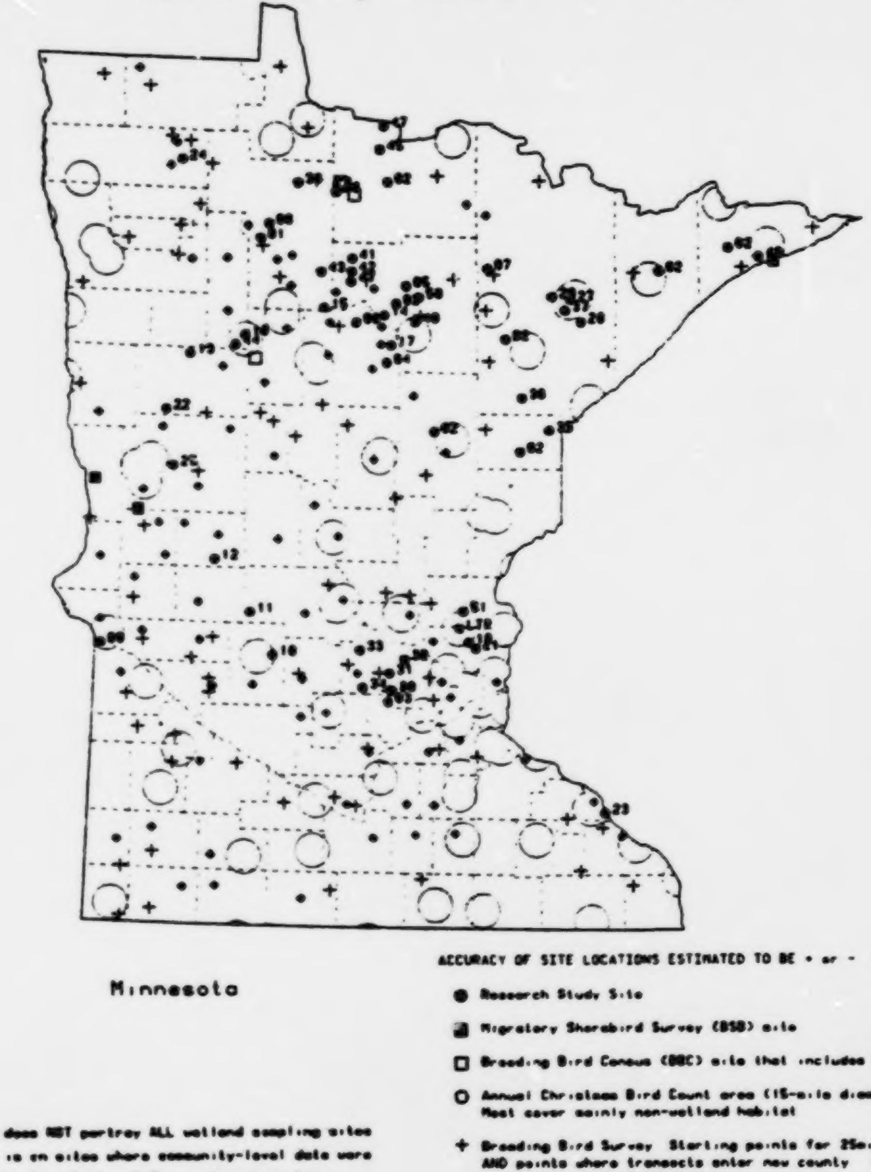
Inland Wetlands Having Biological Community Measurements



Minnesota

ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi.

- ● Research Study Site
- ■ Migratory Shorebird Survey (MSS) site
- ☐ Breeding Bird Census (BBC) site that includes wetland
- ○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat
- + Breeding Bird Survey Starting points for 25mi. transects AND points where transects enter new county. Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

- ◆ State/Federal waterfowl survey

This map does NOT portray ALL wetland sampling sites. Emphasis is on sites where community-level data were collected. See chapter 1 for inclusion criteria.

Sites are referenced by code number to the accompanying state bibliography.

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria   Cartography Jeff Irish   July 1990

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MINNESOTA

Mapped

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MN9-13
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MN18
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MN51
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MNBBS1-
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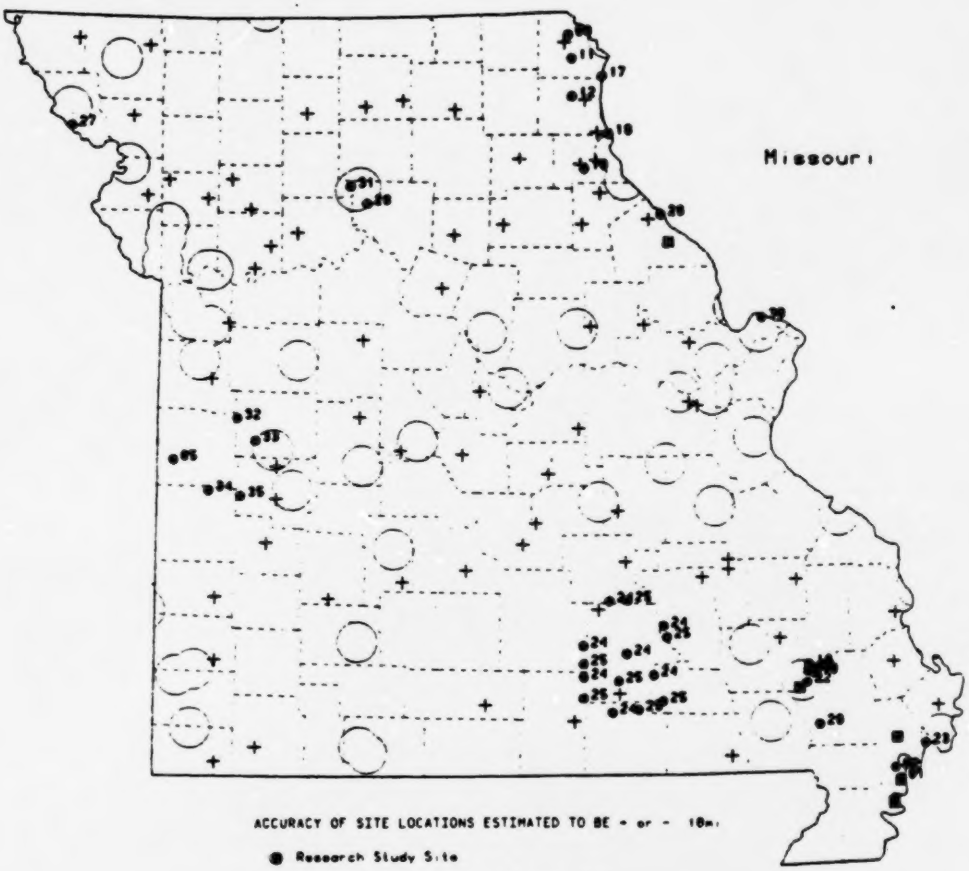
Inland Wetlands Having Biological Community Measurements



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

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AND points where transects enter new county Most cover
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SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

◆ State/Federal waterfowl survey

This map does NOT portray ALL wetland sampling sites
Emphasis is on sites where community-level data were
collected See chapter 1 for inclusion criteria

Sites are referenced by code number to the accompanying
state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1990

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MISSOURI

Mapped

MO1-2
Landin, M.C. 1985. Bird and Mammal Use of Selected Lower Mississippi River Borrow Pits. Ph.D. Diss., Mississippi State Univ., 405 pp. B MA

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MO19
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White, D.C. 1985. Lowland hardwood wetland invertebrate community and production in Missouri. Arch. Hydrobiol. 103(4):509-533. AI

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MO19
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MOBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

MISSOURI (continued)

MOBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

MOBW1-
Missouri Department of Conservation. Unpub. Waterfowl census data. B

MOBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

MOCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Enns, W.R. 1967. Insects Associated with Midwestern Oxidation Lagoons. Terminal Prog. Rep., Univ. Missouri, Columbia, MO. 21 pp. AI

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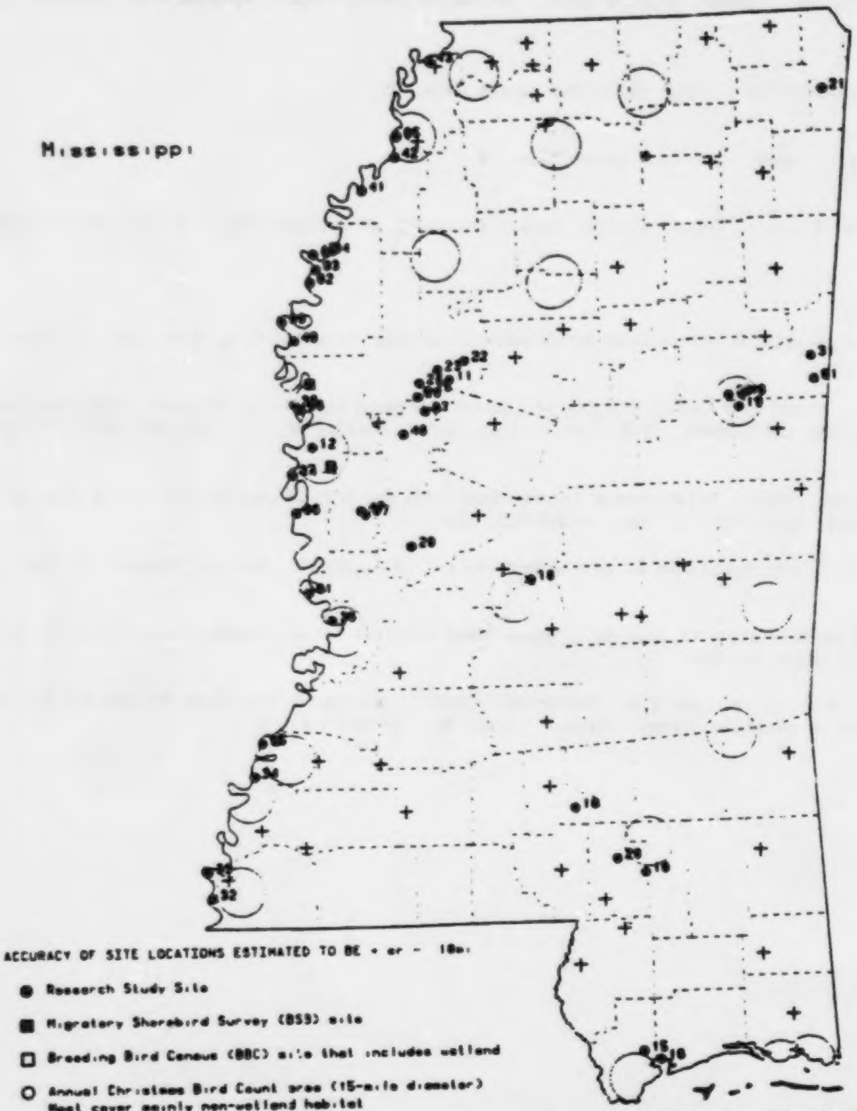
Inland Wetlands Having Biological Community Measurements



ACCUARACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

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SITE LOCATED IN COUNTY SPECIFIC LOCATION(S) NOT PLOTTED

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Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

MISSISSIPPI

Mapped

MS1-5
Landin, M.C. 1985. Bird and Mammal Use of Selected Lower Mississippi River Borrow Pits. Ph.D. Diss., Mississippi State Univ. 405 pp. B MA

MS6-29
Pennington, C.H., H.L. Schramm,Jr., M.E. Potter, and M.P. Farrell. 1980. Aquatic Habitat Studies on the Lower Mississippi River, River Mile 480 to 530. Rep. 5., Environ. Lab. U.S. Army Engr. Waterw. Expt.Stn. Vicksburg, MS. Misc. Paper E-80-1. 101 pp. F

MS6-29
Mathis, D.B., S.P. Cobb, L.G. Sanders, A.D. Magoun, and C.R. Bingham. 1981. Aquatic Habitat Studies on the Lower Mississippi River, River Mile 480 to 530. Rep. 3. Environ. Lab. U.S. Army Engr. Waterw. Expt.Stn., Vicksburg, MS. Misc. Papers E-80-1. 83 pp. F AI

MS 6-29
Schramm, H.L., Jr. and C.H. Pennington. 1981. Aquatic habitat studies on the lower Mississippi River, River Mile 480 to 530. Rep. 6., Environ. Lab. U.S. Army Engr., Waterw. Expt.Stn., Vicksburg, MS. Misc. Paper E-80-1. 74 pp. F

MS6
Pennington, C.H., H.L. Schramm,Jr., M.E. Potter, and M.P. Farrell. 1980. Aquatic Habitat Studies on the Lower Mississippi River, River Mile 480 to 530. Rep. 5., Environ. Lab. U.S. Army Engr. Waterw. Expt.Stn. Vicksburg, MS. Misc. Paper E-80-1. 101 pp. F

MS7
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MS10
Baker, J.A. and S.T. Ross. 1981. Spatial and temporal resource utilization by southeastern cyprinids. Copeia 1981:178-189. F

MS10
Ross, S.T. and J.A. Baker. 1983. The response of fishes to periodic spring floods in a southeastern stream. Amer. Midl. Nat. 109(1):1-15.

MS11
Cooper, C.M. 1987. Benthos in Bear Creek, Mississippi: Effects of habitat variation and agricultural sediments. J. Freshw. Ecol. 4(1):101-113. AI I

MS11
Cooper, C.M. and J.W. Burris. 1984. Bryozoans--possible indicators of environmental quality in Bear Creek, Mississippi. J. Environ. Qual. 13(1):127-130. AI

MS12
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MS13,37
Cobb, S.P. and J.R. Clark. 1981. Aquatic Habitat Studies on the Lower Mississippi River, River Mile 480 to 530. Rep. 2, Environ. Lab. U.S. Army Engr. Waterways Expt.Stn. Vicksburg, MS, Misc. Paper E-80-1. 24 pp. AI F

MS13,37
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MS18-20
Teels, B.M., G. Anding, D.H. Arner, E.D. Noorwood, and D.E. Wesley. 1976. Aquatic plant-invertebrate and waterfowl associations in Mississippi. SE Assoc Game & Fish Comm. 13th Ann. Conf. AI

MISSISSIPPI (continued)

MS22-27
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MS23
Baker, J.A., C.H. Pennington, C.R. Bingham, and L.E. Winfield. 1987. An Ecological Evaluation of Five Secondary Channel Habitats in the Lower Mississippi River. U.S. Army Corps of Engr., Mississippi River Comm., Lower Mississippi River Environ. Prog., Rep. 7. Vicksburg, MS.

MS28
Ross, S.T. and J.A. Baker. 1983. The response of fishes to periodic spring floods in a southeastern stream. Amer. Midl. Nat. 109(1):1-15. F

MS30
Kaminski, R.M. nd. Waterbird Use of "Moist-Soil" Impoundments in Noxubee National Wildlife Refuge, Mississippi. Unpub. Report. Dept. Wildl. and Fisheries, Mississippi S. Univ., 32 pp. B

MS31
U.S. Environmental Protection Agency. 1983. Hydrographic, Water Quality and Biological Studies of Freshwater Canal Systems, South Carolina, Mississippi, and Florida. USEPA, Environ. Serv. Div., Athens, GA. AI

MS32-39
Webb, J.W. and C.V. Klimas. 1988. Vegetation Development on Revetments Along the Lower Mississippi River. U.S. Army Corps of Engr., Mississippi River Commission, Lower Mississippi River Environ. Prog., Rep. 15. Vicksburg, MS.

MS36
Beckett, D.C., C.R. Bingham, and L.G. Sanders. 1983. Benthic macroinvertebrates of selected habitats of the lower Mississippi River. J. Freshw. Ecol. 2(3):247-261. AI

MS37
Conner, J.V., C.H. Pennington, and T.R. Bosley. 1983. Larval Fish Selected Aquatic Habitats on the Lower Mississippi River. Tech. Rep. E-83-4, U.S. Army Engr. Waterw. Expt.Stn. CE, Vicksburg, MS. F

MS40-43
Sigrest, J.M. and S.P. Cobb. 1987. Evaluation of Bird and Mammal Utilization of Dike Systems along the Lower Mississippi River. U.S. Army Corps of Engr., Mississippi River Commission, Lower Mississippi River Environ. Prog. Rep. 10. Vicksburg, MS. 103 pp.

MSBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

MSBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

MSCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Anderson, R.V. and D.M. Day. 1986. Predictive quality of macro-invertebrate - habitat associations in lower navigation pools of the Mississippi River. Hydrobiol. 136:101-112. AI

Arner, D.H., E.D. Norwood, Jr., and B.B. Teels. 1970. A Study of the Aquatic Ecosystem in Two National Waterfowl Refuges in Mississippi. Water Resour. Res. Inst., Mississippi State Univ., MS. 33 pp. AI

Arner, D.H., Robinette, H.R. Frasier, J.E., and M.H. Grey. 1976. Effects of Channelization of the Luxapalila River on Fish, Aquatic Invertebrates, Water Quality and Furbearers. U.S. Fish & Wildl. Serv., Washington, DC, FWS/OBS-76-08. AI MA F I

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Klimas, C.V., C.O. Martin, and J.W. Teaford. 1981. Impacts of Flooding Regime Modification on Wildlife Habitats of Bottomland Hardwood Forests in the Lower Mississippi Valley. U.S. Army Engr. Waterw. Expt.Stn., Rep. # EL-81-13. Vicksburg, MS. 200 pp. I

Klimas, C.V. 1988. Forest Vegetation of the Leveed Floodplain of the Lower Mississippi River. U.S. Army Corps of Engr., Mississippi River Commission, Lower Mississippi River Environ. Prog., Rep. 11. Vicksburg, MS.

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Peterson, M.S. 1987. Ecological and physiological factors affecting the assembly of littoral fish communities along an environmental gradient. Ph.D. Diss., Univ. Southern Mississippi, Hattiesburg, MS. 132 pp.

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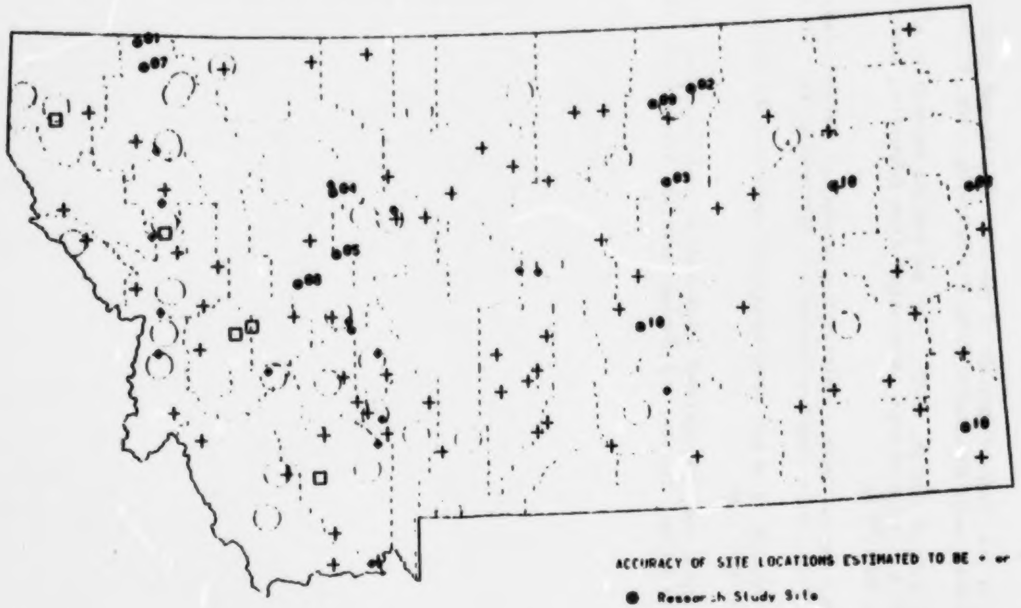
Wiseman, J.B. 1982. A study of the composition, successional relationships, and floristics of Mississippi River floodplain forests in parts of Washington, Bolivar, and Sharkey Counties, Mississippi. Ph.D. Diss., Mississippi St. Univ., MS. 286 pp.

Inland Wetlands Having Biological Community Measurements



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STIE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

MONTANA

Mapped

MT1
Lee, L.C., T.M. Hinckley, and M.L. Scott. 1985. Plant water status relationships among major floodplain sites of the Flathead River, Montana. Wetlands 5:15-34. PW

MT2
Lambing, J.H., W.E. Jones, and J.W. Sutphin. 1988. Reconnaissance Investigation of Water Quality, Bottom Sediment, and Biota Associated with Irrigation Drainage in Bowdoin National Wildlife Refuge and Adjacent Areas of the Milk River Basin, Northeastern Montana. U.S. Geol. Surv., Reston, VA. AI BA I

MT3
Hudson, M.S. 1983. Waterfowl production on three age-classes of stock ponds in Montana. J. Wildl. Manage. 47(1):112-117. B

MT4
Knight, R.R. 1965. Vegetation characteristics and waterfowl useage of a Montana water area. J. Wildl. Manage. 29:782-788. B P

MT5, 6
Elser, A.A. 1968. Fish populations of a trout stream in relation to major habitat zones and channel alterations. Trans. Amer. Fish. Soc. 97:389-397. F I

MT7
Allen, M.L. 1980. Floodplain plant communities of the north fork Flathead River, Montana. Unpub. Report, Nat. Park Serv., Glacier Nat. Park. 98 pp. PW

MT8
Boggs, K.W. 1984. Succession in riparian communities of the lower Yellowstone River, Montana. M.S. Thesis, Montana State Univ., Bozeman, MT, 106 pp. PW

MT9
Gjersing, F.M. 1971. A study of waterfowl production on two rest rotation grazing units in north central Montana. M.S. Thesis, Fish and Wildl. Manage., Montana State Univ., Bozeman, MT. PW

MT10
Smith, R.H. 1953. A study of waterfowl production on artificial reservoirs in eastern Montana. J. Wildl. Manage. 17:276-291. B

MTBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

MTBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

MTBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

MTCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Berg, P.F. 1956. A study of waterfowl broods in eastern Montana with special reference to movements and the relationship of reservoir fencing to production. J. Wildl. Manage. 20(3):253-262.

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MONTANA (continued)

Foote, G.G. 1965. Phytosociology of the bottomland hardwood forests in western Montana. M.S. Thesis, Univ. Montana, Missoula, MT, 140 pp. PW

Garrett, P.A. 1983. Relationships between benthic communities, land use, chemical dynamics, and trophic state in Georgetown Lake, Montana. Ph.D. Diss., Montana St. Univ., Bozeman, MT. 164 pp.

Kessel, S.R. and M.W. Potter. 1980. A quantitative succession model for nine Montana forest communities. Environ. Manage. 4:227-240.

Lee, L.C. 1983. The floodplain and wetland vegetation of two Pacific Northwest river ecosystems. Ph.D. Diss., Univ. Washington, Seattle. 128 pp.

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McBride, J.R. and J. Strahan. 1984. Establishment and survival of woody riparian species on gravel bars of an intermittent stream. Amer. Midl. Nat. 112:235-244. PW

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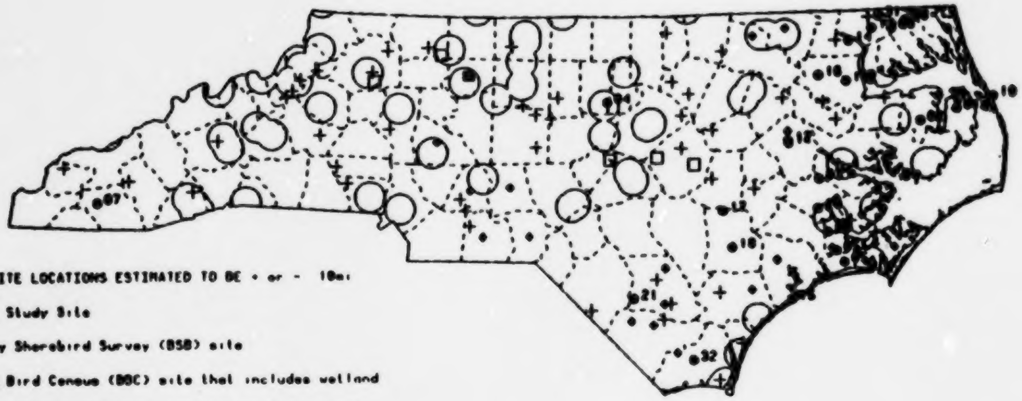
Inland Wetlands Having Biological Community Measurements

North Carolina



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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Rentoria Cartography Jeff Irish July 1990

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NORTH CAROLINA

Mapped

NC1,2
Chescheir, G.M., J.W. Gilliam, R.W. Skaggs, R.G. Broadhead, and R. Lea. 1987. The hydrology and pollutant removal effectiveness of Wetland buffer areas receiving pumped agricultural drainage water. U.S. Geol. Surv. & N. Carolina Water Resour. Res. Inst., Raleigh, NC. P I

NC3
Walker, M.D., R. Sniffen, and W. Sanville. 1985. Fish utilization of aninunidated swamp-stream floodplain. U.S. Environ. Protection Agency, Environ. Res. Lab., Off. Res. & Dev. Corvallis, OR. EPA-600/3-85-046. F

NC4
Morin, P.J. 1984. Odonate guild composition: experiments with colonization history and fish predation. Ecol.65(6):1866-1873. AI

NC5
Christensen, N.L., R.B. Wilbur, and J.S. McLean. 1988. Soil-vegetation Correlations in the Pocosins of Croatan National Forest, North Carolina. Biol. Rep. 88(28) U.S. Fish Wildl. Serv., Washington, D.C. 98 pp. P

NC6
Atchue, A., III, F.P. Day, Jr., and H.G. Marshall. 1983. Algal dynamics and nitrogen and phosphorus in a cypress stand in the seasonally flooded Great Dismal Swamp. Hydrobiol. 106:115-122. A

NC7
Meyer, J.L. and C. Johnson. 1983. Influence of elevated nitrate concentration on rate of leaf decomposition in a stream. Freshw. Biol. 13(2):177-183. D

NC8
Megonigal, J.P. and F.P. Day, Jr. 1988. Organic matter dynamics in four seasonally flooded forest communities of the Dismal Swamp. Amer. J. Bot. 75(9):1334-1343. SO PW

NC11,12
Brinson, M.M. 1977. Decomposition and nutrient exchange of litter in an alluvial swamp forest. Ecol. 58:601-609. D

NC13
Pardue, G.H., M.T. Huish, and H.R. Perry, Jr. 1975. Ecological Studies of Two Swamp Watersheds in Northeastern North Carolina. A Prechannelization Study. NC Water Resour. Res.Inst., Raleigh, Rep. #UNC-WRRI-75-105, (NTIS Pb-242 126/Ost.) 472 pp. P MA B

NC13
Pardue, G.B. and M.T. Huish. 1981. An evaluation of methods for collecting fishes in swamp streams. pp. 282-290 in L.A. Krumholz. The Warmwater Streams Symposium. Amer. Fish Soc., Bethesda, MD. T F

NC14
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NC15
Lenat, D. 1985. Mill Creek Survey, August, 1985. North Carolina Div. Environ. Manage., 20 pp.

NC16
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NORTH CAROLINA (continued)

NC16,24
Kuenzler, E.J. 1987. Impacts of sewage effluent on tree survival, water quality, and nutrient removal in Coastal Plain swamps. UNCWRRI-87-235, Water Resources Res. Inst., North Carolina State Univ., Raleigh, NC.

NC17
Penrose, D.P. 1986. Investigation of the Mt. Olive Pickle Plant and the Mt. Olive WWTP. North Carolina Div. Environ. Manage., Raleigh, 19 pp.

NC17
Penrose, D.P. 1987. Bioassessment of the Cates Pickle Company effluent, Duplin County. North Carolina Div. Environ. Manage., Raleigh, 8 pp. AI

NC18
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NC19
MacPherson, T.F. 1987. Nags Head Woods - Pond Study. North Carolina Div. Environ. Manage., Raleigh. 18 pp.

NC20
Lenat, D. 1989. Effect of discharge from Clarks Quarry (Martin Marietta) on Caswell Branch, Craven County. North Carolina Div. Environ. Manage., Raleigh.

NC21
Penrose, D. 1989. Biological Survey of Raft Swamps near the City of Lumberton's Abandoned Landfill. North Carolina Div. of Environ. Manage., Raleigh.

NC22-30
Woodwell, G. 1956. (Unpub. data), In: N.L. Christensen, R.B. Wilbur, J.S. McLean. 1988. Soil-Vegetation Correlations in the Pocosins of Croatan National Forest, North Carolina. U.S. Fish and Wildl. Serv., Raleigh. Biol. Rep. 88(28).

NC31
Carter, V., M.K. Garrett, and P.T. Gammon. 1988. Wetland boundary determination in the Great Dismal Swamp using weighted averages. Water Resour. Bull., 24(2):297-306.

NC32
Kologiski, R.L. 1977. The phytosociology of the Green Swamp, North Carolina. NC Agric. Expt. Stn., Tech. Bull. No. 250. Raleigh, NC.

NC33
Huish, M.T. and G.B. Pardue. 1978. Ecological studies of one channelized and two unchannelized wooded coastal swamp streams in North Carolina. U.S.Fish and Wildl. Serv., FWS 10BS-78/85. 72 pp. I

NCBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

NCBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

NCBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

NCBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

NCCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

NORTH CAROLINA (continued)

Not Mapped

Briscoe, C.B. 1957. Diameter growth and effects of flooding on certain bottomland forest trees. Ph.D. Diss., Duke Univ., Raleigh, N.C.

Flinchum, D.M. 1977. Lesser vegetation as indicators of varying moisture regimes in bottomland and swamp forests of northeastern North Carolina. Ph.D. Diss., North Carolina St. Univ., Raleigh, NC. 110 pp.

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Maki, T.E., D.W. Hazel, and A.J. Weber. 1980. Effects of Stream Channelization on Bottomland and Swamp Forest Ecosystems. North Carolina State Univ., Raleigh, NC. 66 pp. (NTIS Pb-269 021/2St). I MA B P

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Sniffen, R.P. 1981. Benthic invertebrate production during seasonal inundation of a floodplain swamp. Ph.D. Diss., Univ. North Carolina, Chapel Hill, NC. 189 pp.

Tarplee, W.H., Jr. 1975. Studies of the Fish Populations in Two Eastern North Carolina Swamp Streams. North Carolina State Univ., Dept. of Zool., Raleigh, NC. NTIS Pb-269 104/6St. F

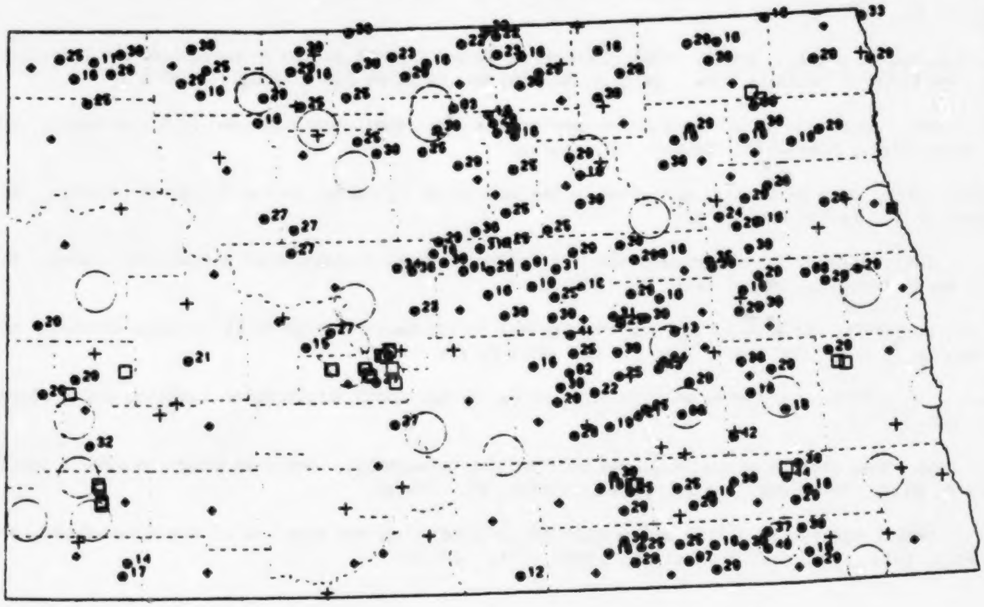
Tarplee, W.H., Jr. 1979. Estimates of fish populations in two northeastern North Carolina swamp streams. Brimleyana 1:99-11. F

Teate, J.L. 1968. Some effects of environmental modification on vegetation and tree growth in a North Carolina pocosin. Ph.D. Diss., North Carolina St. Univ., Raleigh, NC. 119 pp.

Walker, J.L. 1985. Species diversity and production in pine-wiregrass savannas of the Green Swamp, North Carolina. Ph.D. Diss., Univ. North Carolina, Chapel Hill. 260 pp.

Inland Wetlands Having Biological Community Measurements

North Dakota



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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

NORTH DAKOTA

Mapped

ND1
Faanes, C.A. 1982. Avian Use of Sheyenne Lake and Associated Habitats in Central North Dakota. U.S. Fish & Wildl. Serv. Resour. Pub. 144 Washington, D.C. 24pp. B

ND2
Swanson, G.A. and M.I. Meyer. 1977. Impact of fluctuating water levels on feeding ecology of breeding Blue-Winged Teal. J. Wildl. Manage. 41(3):426-433. B I

ND3-4
Hammond, M.C. and D.H. Johnson. 1984. Effects of Weather on Breeding Ducks in North Dakota. U.S. Fish & Wildl. Serv. Tech. Rep. #1. B

ND5
Voorhees, L.D. and J.F. Cassel. 1980. Highway right-of-way: mowing versus succession as related to duck nesting. J. Wildl. Manage. 44(1):155-163. B I

ND6
Swanson, G.A., V.A. Adomaitis, F.B. Lee, J.R. Serie, and J.A. Shoesmith. 1984. Limnological conditions influencing duckling use of saline lakes in south-central North Dakota. J. Wildl. Manage. 48(2):340-349. B

ND7-11
Klett, A.T., T.L. Shaffer, and D.H. Johnson. 1988. Duck nest success in the Prairie Pothole region. J. Wildl. Manage. 52(3):431-440. B R

ND12
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ND13
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ND16
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ND17
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ND18
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ND19
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ND20
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ND24
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ND28
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ND29
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ND31
Weinhold, C.E. and A.G. van der Valk. 1988. The impact of duration of drainage on the seed banks of northern prairie wetlands. Can. J. Bot. 67:1878-1884.

ND32
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ND33
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ND43
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ND43
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ND44
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NDBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

NDBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

NDBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

NDCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

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Johnson, W.C., R.L. Burgress, and W.R. Keammerer. 1976. Forest overstory vegetation and environment on the Missouri River floodplain in North Dakota. Ecol. Monogr. 46:59-84. PW

Kaloupek, L. 1972. A taxonomic and distributional study of the aquatic vascular plants of northeastern North Dakota. M.S. Thesis, Univ. North Dakota, Grand Forks, ND. P

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Larson, G.E. 1979. The aquatic and wetland vascular plants of North Dakota. Ph.D. Diss., North Dakota State Univ., Fargo, ND. 459 pp. P

Malterer, T.J., A.J. Duxbury, and J.L. Richardson. 1987. Soil character of three calcareous fens in North Dakota and Minnesota. N. Dakota Acad. Sci. 41:65. P

Reily, P.W. and W.C. Johnson. 1982. The effects of altered hydrologic regime on tree growth along the Missouri River in North Dakota. Can. J. Bot. 60:2410-2423. PW

Rossiter, J.A. and Crawford R.D. 1981. Evaluation of Constructed Ponds as a Means of Replacing Natural Wetland Habitat Affected by Highway Projects in North Dakota. State Study (2)-79(A), Biol. Dept., Univ. North Dakota, Grand Forks, ND. 169 pp. B

Stewart, R.E. and H.A. Kantrud. 1973. Ecological distribution of breeding waterfowl populations in North Dakota. J. Wildl. Manage. 37:39-50.

Swanson, G.A. 1977. Diel food selection by Anatinae on a waste stabilization system. J. Wildl. Manage. 41(2):226-231. B I

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Wali, M.K. 1976. Comparative studies of some inland saline aquatic ecosystems in North Dakota. North Dakota Water Resources Res. Inst. Rep. No. WI-221-033-76, Fargo, ND. P

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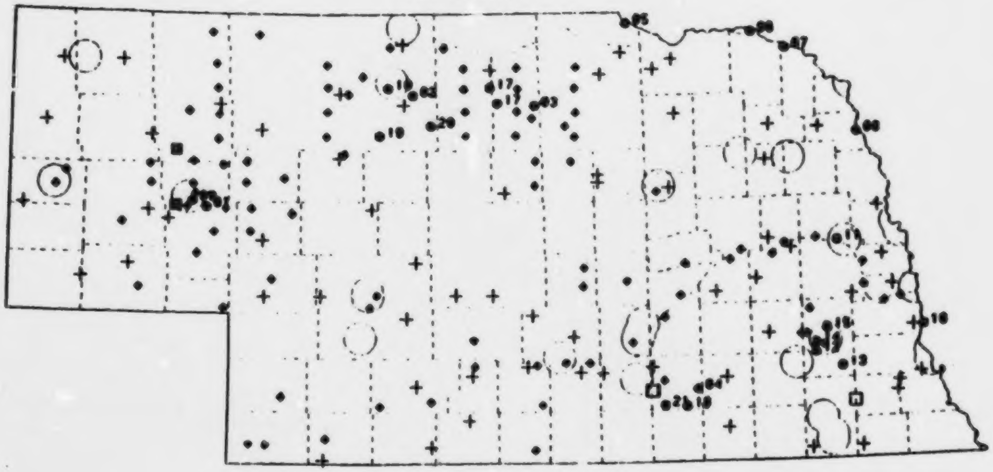
Inland Wetlands Having Biological Community Measurements

Nebraska



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

- ● Research Study Site
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- ☐ Breeding Bird Census (BBC) site that includes wetland
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- + Breeding Bird Survey Starting points for 25mi transects AND points where transects enter new county Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

- ● State/Federal waterfowl survey

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NEBRASKA

Mapped

NE1-4
Erickson, N.E. and D.M. Leslie, Jr. 1987. Soil-vegetation correlations in the Sandhills and Rainwater Basin wetlands of Nebraska. U.S. Fish & Wildl. Serv. Biol. Rep. 87(11):73. P

NE5-8
Kallemeyn, L.S. and J.F. Novotny. 1977. Fish and fish food organisms in various habitats of the Missouri River in South Dakota, Nebraska and Iowa. U.S. Fish & Wildl. Serv., Washington, D.C. FWS/OBS-77/25. AI F

NE9-13
Mahoney, D.L. 1977. Species richness and diversity of aquatic vascular plants in Nebraska with special reference to water quality parameters. M.S. Thesis, Univ. of Nebraska-Lincoln. 38 pp.

NE14,15
Ducey, James E. 1987. Biological features of saline wetlands in Lancaster County, Nebraska. Trans. Nebraska Acad. Sci. 15:5-14.

NE16
Golden, D.R. 1987. An ichthyological survey of Weeping Water Creek, Nebraska. Trans. Nebraska Acad. Sci. 15:15-22.

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Maret, T.R. 1988. A water-quality assessment using aquatic macro-invertebrates from streams of the Long Pine Creek watershed in Brown County, Nebraska. Trans. Nebraska Acad. Sci. 69-84 pp.

NEBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

NEBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

NEBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

NEBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

NECBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

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Inland Wetlands Having Biological Community Measurements

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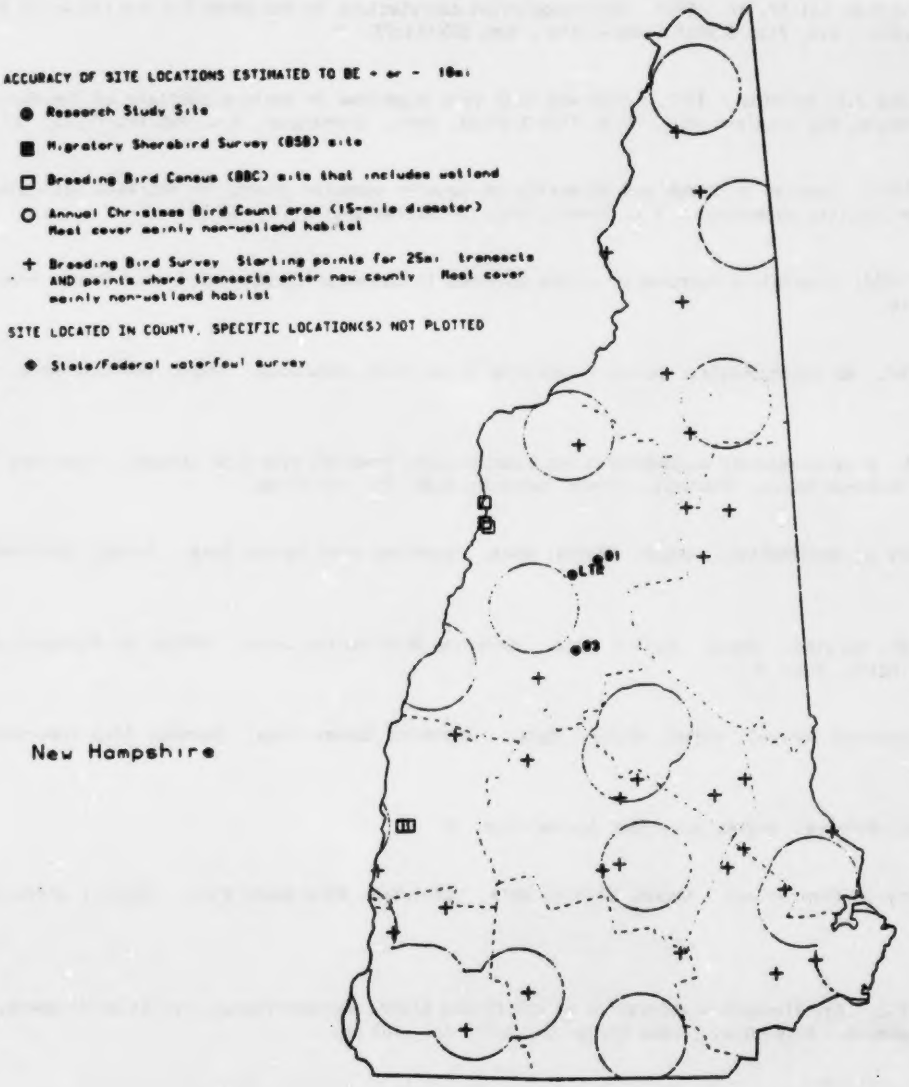

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USEPA Environmental Research Laboratory, Corvallis, Oregon

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NEW HAMPSHIRE

Mapped

NH1
Moeller, Robert E. 1975. Hydrophyte biomass and community structure in a small, oligotrophic New Hampshire lake. Verh. Inter. Verein. Limnol. 19:1004-1012. PM

NH2
Fahey, T.J. et al. In Process. Long Term Environmental Research Wetland Site: Mirror Lake.

NH3
Glime, J.M. and R.M. Clemons. 1972. Species diversity of stream insects on Fontinalis spp. compared to diversity on artificial substrates. Ecol. 53(3):458-464. PM AI

NHBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

NHBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

NHBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

NHCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

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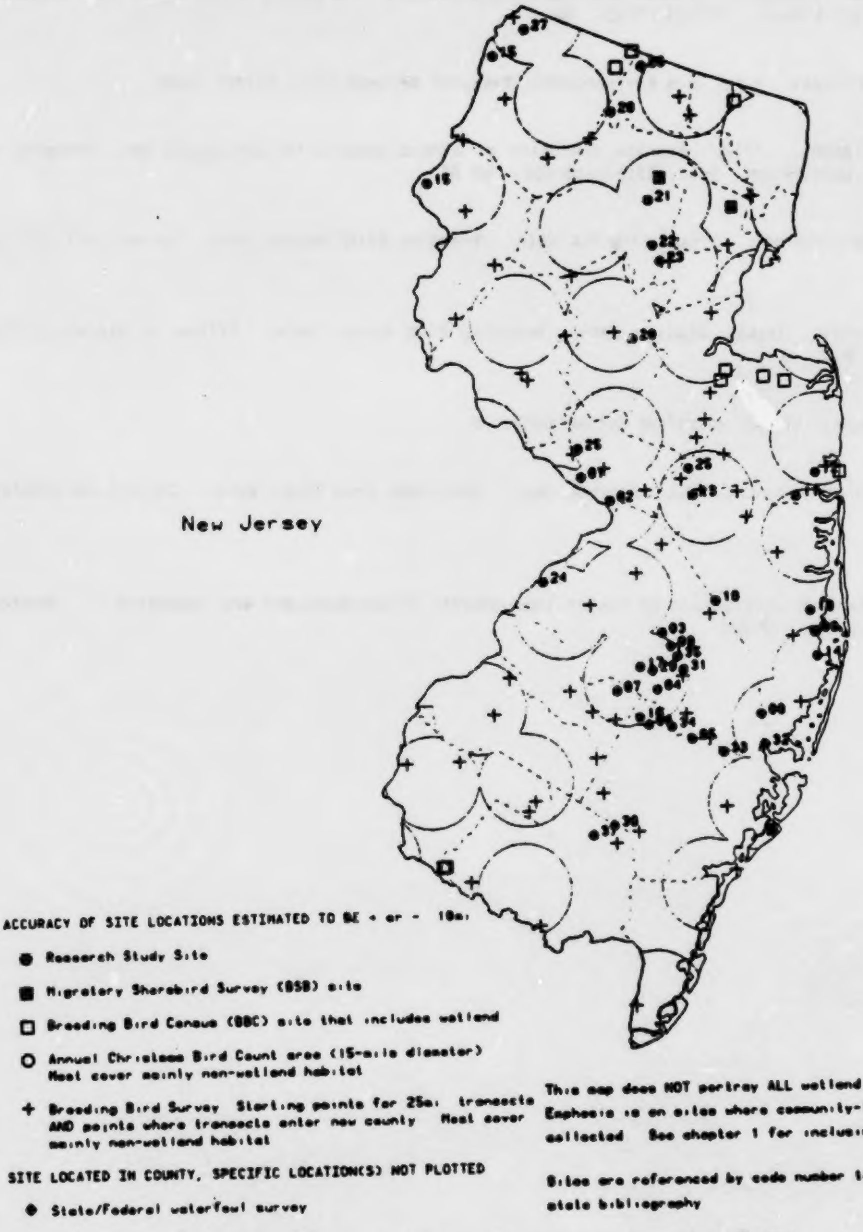
Inland Wetlands Having Biological Community Measurements



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NEW JERSEY

Mapped

NJ1
Garie, H.L. and A. McIntosh. 1986. Distribution of benthic macroinvertebrates in a stream exposed to urban runoff. Water Resour. Bull. 22(3):447-454. AI I

NJ3
Ehrenfeld, J.G. and M. Gulick. 1981. Structure and dynamics of hardwood swamps in the New Jersey Pine Barrens: contrasting patterns in trees and shrubs. Amer. J. Bot. 68(4):471-481.

NJ3-6
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NJ7-13
Ehrenfeld, J.G. 1983. The effects of changes in land use on swamps of the New Jersey Pine Barrens. Biol. Cons. 25:353-375. P R

NJ16-19
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NJ19
Cole, C.A. 1988.

NJ20
Buchholz, K. 1981. Effects of minor drainage on woody species distributions in a successional floodplain forest. Can. J. For. Res. 11:671-676. PW I

NJ21
Jervis, R.A. 1969. Primary production in the freshwater marsh ecosystem of Troy Meadows, New Jersey. Bull. Torrey Bot. Club 96(2):209-231. P

NJ22
Scott, D. and L. Bush. 1969. A study of a pond in the Great Swamp. Drew Univ., Madison, NJ.

NJ22
Gatter, R. 1986. A survey of the benthic macroinvertebrates of the Great Swamp National Wildlife Refuge and its immediate environs Morris County, New Jersey. M.S. Thesis, Graduate Program in Ecology. Rutgers Univ., New Brunswick, NJ. AI

NJ24
Kaminsky, M., P. Scelsi, C. Kanakis, and D. Fanz. 1986. Route 130, Section 9F Rancocas Creek bridge: site III Wetland replacement. N.J. Dept. of Trans., Bureau of Environ. Analysis, Trenton, NJ. P

NJ25
Scelsi, P. nd. Small mammal and bird utilization of New Jersey highway interchanges containing wetland habitat. NJ Dept. of Transportation, Trenton. MA B

NJ25
Schneider, J.P. and J.G. Ehrenfeld. 1987. Suburban development and cedar swamps: Effects on water quality, water quantity, and plant community composition. pp. 271-288. In: A.D. Laderman (ed.). Atlantic White Cedar Wetlands, Westview Press, Inc. PH

NJ26-27
Karlin, E.F. 1985. The vegetation of the low-shrub bogs of northern New Jersey and adjacent New York: ecosystems at their southern limit. Bull. Torrey Bot. Club 112(4):436-444.

NJ26-28
Andrus, R. E. 1986. Sphagnum vegetation of the low shrub bogs of northern New Jersey and adjacent New York. Bull. Torrey Bot. Club 113(3):281-287.

NEW JERSEY (continued)

NJ29-34
Hastings, R.W. 1984. The fishes of the Mullica River, a naturally acid water system of the New Jersey Pine Barrens. Bull. New Jersey Acad. Sci. 29(1):9-23. F

NJ35
Brush, T. 1988. Foliage arthropods of the New Jersey Pine Barrens: Seasonal variation in abundance in different plant taxa. Bull. New Jersey Acad. Sci. 33(1):1-6.

NJ36
O'Herron, J.C., II and R.G. Arndt. 1987. Fish Studies in the Manumuskin River Drainage Basin - and Portions of the Maurice River and Manantico Creek, Maurice River Township, Cumberland County, New Jersey. Herptological Associates, Inc., Environmental Consultants. HA File No. 87.01-B.

NJ36
Sutton, C.C., R. Barber, and J. Dowdell. 1987. An Inventory and Habitat Assessment of the Birds of the Manumuskin River Drainage System and Portions of the Adjacent Maurice River in Cumberland County, New Jersey. Herptological Associates, Inc., Environ. Consultants. HA File No. 87.01-A.

NJ36
Zappalorti, R.T. and R.D. Barber. 1987. Mammalogical and Herpetological Studies in the Manumuskin River Drainage Basin in Cumberland and Atlantic Counties, New Jersey between 1986 and 1987. Herptological Associates, Inc., Environmental Consultants. HA File No. 87.01-C.

NJBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

NJBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

NJBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

NJBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

NJCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Atkin, D. 1980. The age structures and origins of trees in a New Jersey floodplain forest. Ph.D. Diss., Princeton Univ., Princeton, NJ. 119 pp.

Ehrenfeld, J.G. and J.P. Schneider. 1987. The effects of suburban development on water quality and vegetation of cedar swamps. Center for Coastal and Environ. Studies, Rutgers Univ., New Brunswick, NJ. P I

Frye, R.J., II and J.A. Quinn. 1979. Forest development in relation to topography and soils on a floodplain of the Raritan River, New Jersey. Bull. Torrey Bot. Club 106:334-345. PW

Lomax J.L. 1982. Wildlife use of mineral industry sites in the coastal plains of New Jersey. pp. 115-121 In: W.D. Svedarsky and R.D. Crawford (eds.). Wildlife Values of Gravel Pits. Proc. Symp., Minnesota Agric. Exp. Stn. Misc. Publ. 17.

Lynn, L.M. and E.F. Karlin. 1985. The vegetation of the low-shrub bogs of Northern NJ and adjacent NY: Ecosystems at their southern limit. Bull. of the Torrey Bot. Club 112:436-444. P

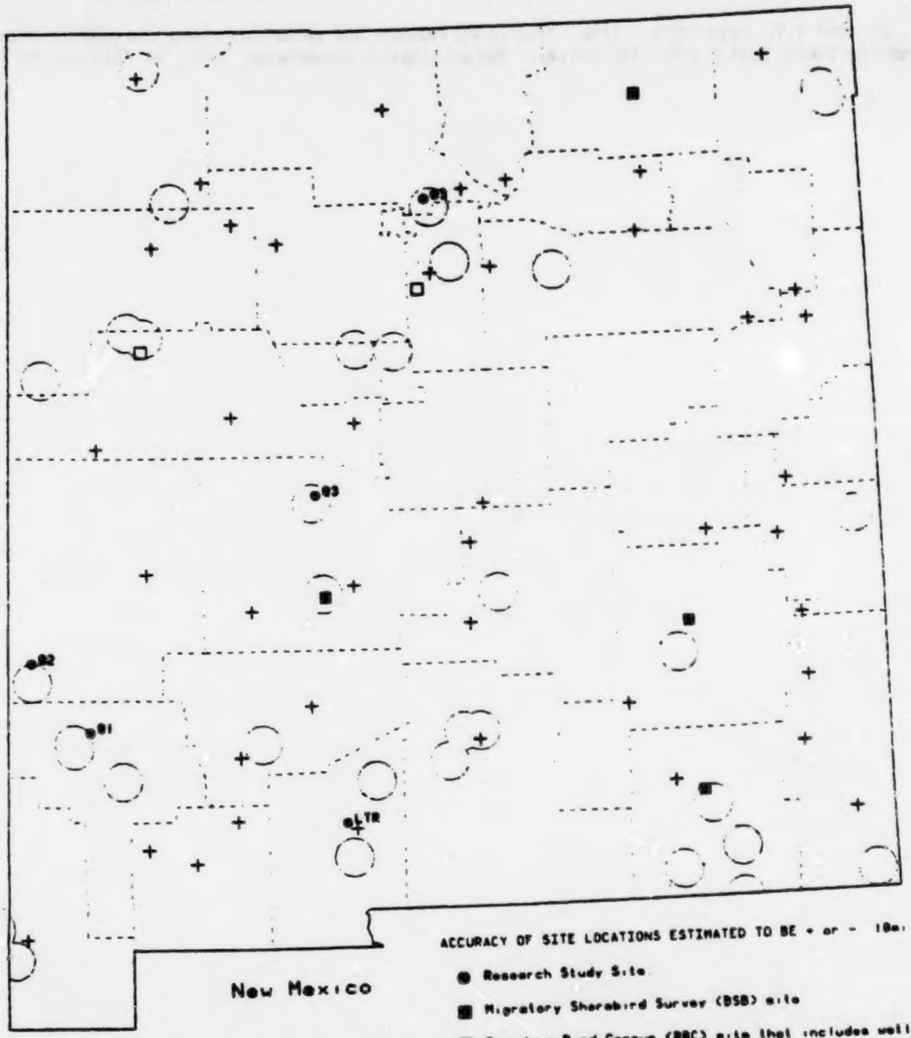
Schneider, J.P. 1988. The effects of suburban development on the hydrology, water quality and community structure of *Chamaecyparis thyoides* wetlands in the New Jersey Pinelands. Ph.D. Diss., Rutgers Univ. P

NEW JERSEY (continued)

Smith, R.F. 1960. An ecological study of an acid pond in the New Jersey coastal plain. Ph.D. Diss., Rutgers Univ., New Brunswick, NJ. 197 pp.

Sutton, C.C., Jr. and R.T. Zappalorti. 1988. Wintering Raptors and Waterfowl along the Maurice River on the Delaware Bayshore, Cumberland County, New Jersey. Herpetological Associates, Inc., HA File No. 87.44. N

Inland Wetlands Having Biological Community Measurements



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Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish

NEW MEXICO

Mapped

NM1, 2
Dick-Peddie, W.A., J.V. Hardesty, E. Muldavin, and B. Sallach. 1987. Soil-vegetation correlations on the riparian zones of the Gila and San Francisco Rivers in New Mexico. Biol. Rep. 87(9). U.S. Fish & Wildl. Serv., Washington, D.C. 30 pp. P

NM3
Hink, V.C. and R.D. Ohmart. 1984. Middle Rio Grande Biological Survey. U.S. Army Corps. of Engr., Contract No. DAC W47-81-C-0015. B

NM4, LTR
Schlesinger, W., et al. In Process. Long Term Environmental Research Wetland Site: Jornada LTER Site. Dept. of Bot., Duke Univ., Durham, NC. P

NMBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

NMBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

NMBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

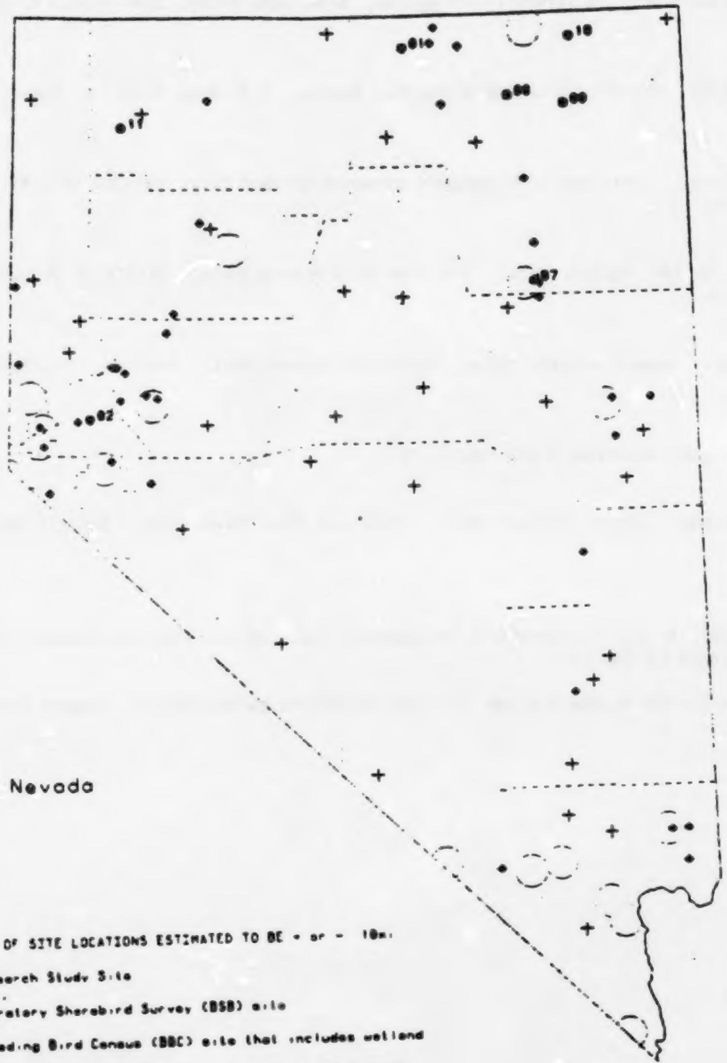
NMCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Szaro, R.C. and J.N. Rinne. 1988. Ecosystem approach to managment of southwestern riparian communities. Tran. N. Amer. Wildl. Nat. Resour. Conf. 53:502-511.

Szaro, R.C. 1989. Riparian forest and scubland community types of Arizona and New Mexico. Desert Plants 9:70-138.

Inland Wetlands Having Biological Community Measurements



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USEPA Environmental Research Laboratory, Corvallis, Oregon

July 1988

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish

NEVADA

Mapped

NV1
Platts, W.S. 1985. The effects of large storm events on basin-range riparian stream habitats.

NV2
Nachlinger, J.L. 1988. Soil-Vegetation Correlations in riparian and emergent wetlands, Lyon County, Nevada. U.S. Fish & Wildl. Serv. Biol. Rep. 88(17) U.S. Fish & Wildl. Serv., Washington, D.C. 40pp. P

NV7
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NV7
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NV7
Young, J.A., R.A. Evans, B.A. Roundy, and J.A. Brown. 1986. Dynamic landforms and plant communities in a pluvial desert rodent communities: Habitats and environmental complexity. Ecol. 50:558-572.

NV7
Young, J.A., R.A. Evans, B.A. Roundy, and J.A. Brown. 1986. Dynamic landforms and plant communities in a pluvial lake basin. Great Basin Nat. 46(1):1-32.

NV8-11
Jensen, S., R. Ryel and W.S. Platts. 1989. Classification of riverine/riparian habitat and assessment of nonpoint source impacts north fork Humboldt River Nevada. USDA Forest Service, Intermountain Research Station, Fisheries Unit, Boise, ID.

NVBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

NVBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

NVCBC1-
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Not Mapped

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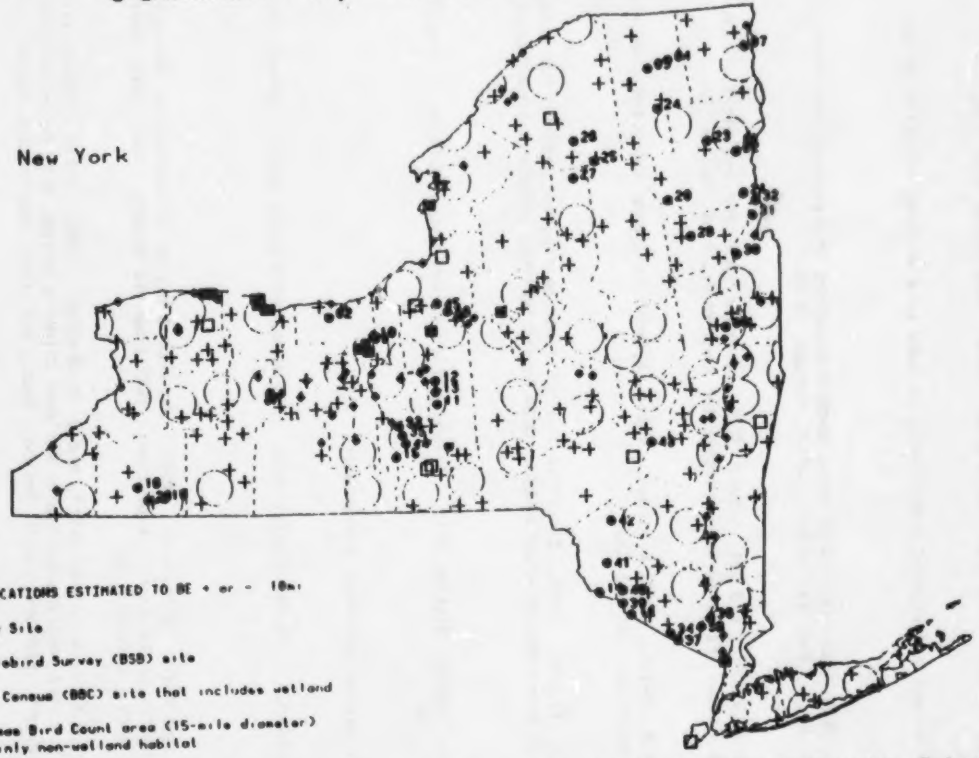
Inland Wetlands Having Biological Community Measurements

New York



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

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Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

July 1988

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish

NEW YORK

Mapped

NY1
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NY2
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NY2-4
Malecki, R.A. and J.D. Sullivan. 1987. Assessment of an agricultural drainage improvement program in New York State. J. Soil Water Conserv. 42:271-274. I P

NY2-4
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NY7
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NY8
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NY9
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NY9
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NY9-12
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NY13
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NY14
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NY15
Delucchi, C.M. 1987. Comparison of community structure among streams with different temporal flow regimes. Can. J. Zool. 66:579-586. AI

NY16
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NY18-19
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NY20-28
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NY29
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NY30-32
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NY33
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NY34
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NY36-42
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NY39-42
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NY43
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NY45
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NY46
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NY47
State University of New York at Oneonta, Biology Dept. nd. Annual Reports 1968 to Date. Biol. Field Stn., Cooperstown, NY. PM AI

NY48
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NYBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

NYBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

NEW YORK (continued)

NYBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

NYBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

NYCBC1-
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NEW YORK (continued)

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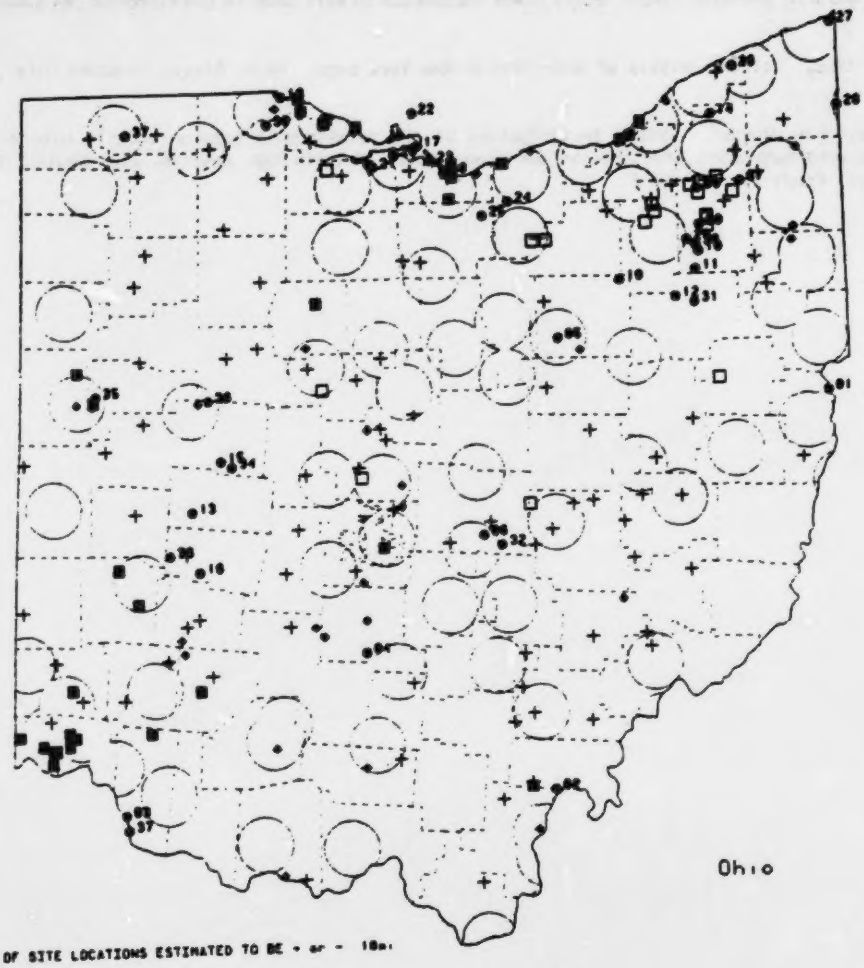
Inland Wetlands Having Biological Community Measurements



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USEPA Environmental Research Laboratory, Corvallis, Oregon

July 1988

Data Compilation Paul Adamus and Robin Rentorio Cartography Jeff Irish

OHIO

Mapped

OH1-3
Van Hassel, J.H., R.J. Reash, and H.W. Brown. 1988. Distribution of upper and middle Ohio River fishes, 1973-1975: I. Associations with water quality and ecological variables. J. Freshw. Ecol. 4(4):441-458. F

OH1-3
Reash, R.J. and J.H. Van Hassel. 1988. Distribution of upper and middle Ohio River fishes, 1973-1985: II. Influence of zoo-geographic and physiochemical tolerance factors. J. Freshw. Ecol. 4(4):459-476. F

OH4-16
Gerono, R.J. and D.B. MacLean. 1988. Caddisflies (Trichoptera) of Ohio wetlands as indicated by light-trapping. Ohio J. Sci. 88(4):143-151. AI

OH17
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OH18,37,38
Tramer, E.J. and P.M. Rogers. 1973. Diversity and longitudinal zonation in fish populations of two streams entering a metropolitan area. Amer. Midl. Nat. 90(2):366-374. F I

OH19
Loveland, D.G. and I.A. Ungar. 1983. The effect of nitrogen fertilization on the production of halophytes in an inland salt marsh. Amer. Midl. Nat. 109(2):346-354. I PE

OH20
Farney, R.A. 1982. Vegetation changes in a Lake Erie marsh (Winous Point, Ottawa County, Ohio) during high water years. Ohio Acad. Sci. 82:103-107. PE

OH21
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OH22
Stuckey, R.L. 1971. Changes of vascular aquatic flowering plants during 70 years in Put-In-Bay Harbor, Lake Erie, Ohio. Ohio J. Sci. 71(6):321-342. PM TS

OH23-36
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OH37
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OH37,38
Tramer, E.J., and P.M. Rogers. 1973. Diversity and longitudinal zonation in fish populations of two streams entering a metropolitan area. Amer. Midl. Nat. 90(2):366-374. F I

OH38
Reeder, B.C. and W.J. Mitsch. 1989. Seasonal patterns of planktonic and macrophyte productivity of a freshwater coastal wetland. pp. 49-55 IN: Wetlands of Ohio's Coastal Lake Erie: A Hierarchy of Systems. Environ Sci. Prog., School of Nat. Resources, Ohio State Univ., Columbus, OH.

OH39
Stuckey, R.L. and W.A. Wentz. 1969. Effect of industrial pollution on the aquatic and shore angiosperm flora in the Ottawa River, Allen and Putman Counties, Ohio. Ohio J. Sci.69:226-242.

OHBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

OHIO (continued)

OHBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

OHBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

OHBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

OHCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

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Cooke, G.D. 1980. Lake level drawdown as a macrophyte control technique. Water Resour. Bull. 16(2):317-322. PM I

Hardin, E.D. 1982. Patterns in floodplain herbaceous vegetation and some aspects of the population biology of Populus deltoides on the Hocking River, Ohio. Ph.D. Diss., Ohio Univ., Athens, OH. 279 pp.

Harter, R.D. 1966. The effect of water levels on soil chemistry and plant growth of the Magee Marsh Wildlife Area. Ohio Game Monogr., No. 2, Ohio Div. Wildl., Columbus, OH, 36 pp. P

Hoffman, W.S. 1985. The Fishes of Old Women Creek Estuary. OWC Tech. Rep. #4, ODNR, Div. of Nat. Areas and Preserves. F

Hufford, T.L. 1972. Analyses of seasonal and areal distribution patterns of diatom taxa of Cedar Bog, Champaign County, Ohio. Ph.D. Diss., Ohio St. Univ., Columbus, OH. 155 pp.

Millie, D. 1979. Periphytic algae in southwestern Lake Erie marshes. M.S. Thesis, Bowling Green St. Univ., Bowling Green, OH. A

Riley, T.Z. 1989. Effects of wetland water level manipulation on macroinvertebrate abundance during the waterfowl breeding season. Ph.D. Dissertation, Ohio St. Univ., Columbus, OH. AI

Rotenberry, J.T., E.E. Emmons, and C.H. Hardman. 1987. Use of Backwater Marsh Areas by Fish Populations in Old Women Creek and Surrounding Lake Erie Prior to Highway Construction. Final report submitted to Sanctuary Programs Div.- NOAA/NOS, and ODNR, Div. of Natural Areas and Preserves. 26 pp. F

Rotenberry, J.T., T.M. Bergin, and B.S. Steiner. 1989. The Influence of Highway Construction on the Use of Backwater Marsh Areas by Fish Populations in Old Women Creek and Surrounding Lake Erie.Final report submitted to NEMD, NOAA/NOS, and ODNR, Div. of Nat. Areas. 22 pp. F

Williams, H.N. 1962. Pollen analysis of two central Ohio bogs. Ph.D. Diss., Ohio St. Univ., Columbus, OH. 72 pp.

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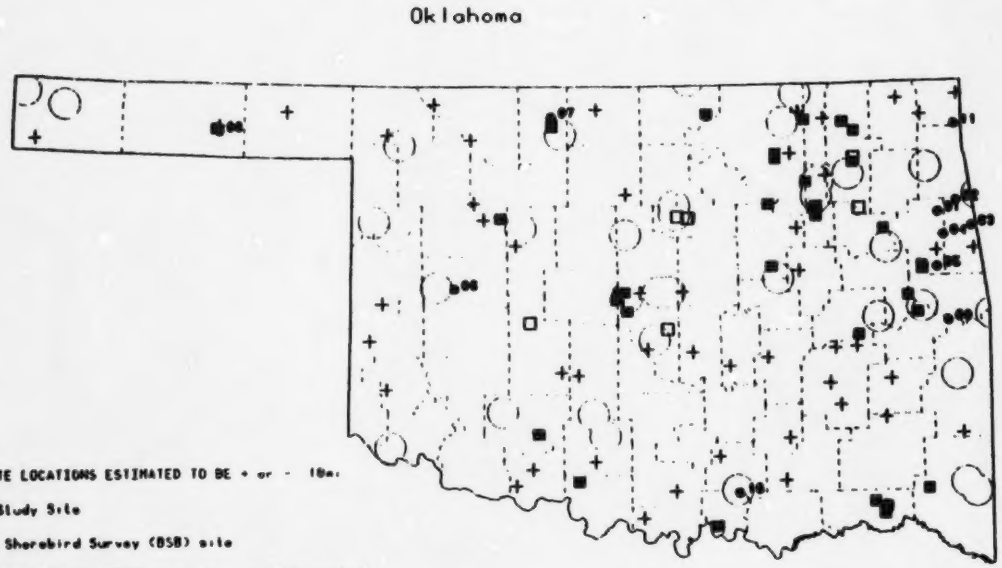
Inland Wetlands Having Biological Community Measurements



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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

OKLAHOMA

Mapped

OK1-5
Felley, J.D. and L.G. Hill. 1983. Multivariate assessment of environmental preferences of cyprinid fish of the Illinois River, Oklahoma. Amer. Midl. Nat. 109(2):209-221. F

OK6-10
Heitmeyer, M.E. and P.A. Vohs, Jr. 1984. Distribution and habitat use of waterfowl wintering in Oklahoma. J. Wildl. Manage. 48(1):51-62. B

OK11
Erickson, N.E. and D.M. Leslie, Jr. 1988a. Impacts of the rule curve change on shoreline vegetation and wildlife around Grand Lake, Oklahoma. Prepared for the Benham-Holway Power Group, Tulsa, OK. 20 pp.

OK11
Erickson, N.E. and D.M. Leslie, Jr. 1988b. Shoreline vegetation and general wildlife values around Grand Lake, Oklahoma. Prepared for the Benham-Holway Power Group, Tulsa, OK. 70 pp.

OKBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

OKBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

OKBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

OKBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

OKCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

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Hannan, H.H. and T.C. Dorris. 1970. Succession of a macrophyte community in a constant temperature river. Limnol. Oceanogr. 15:442-453.

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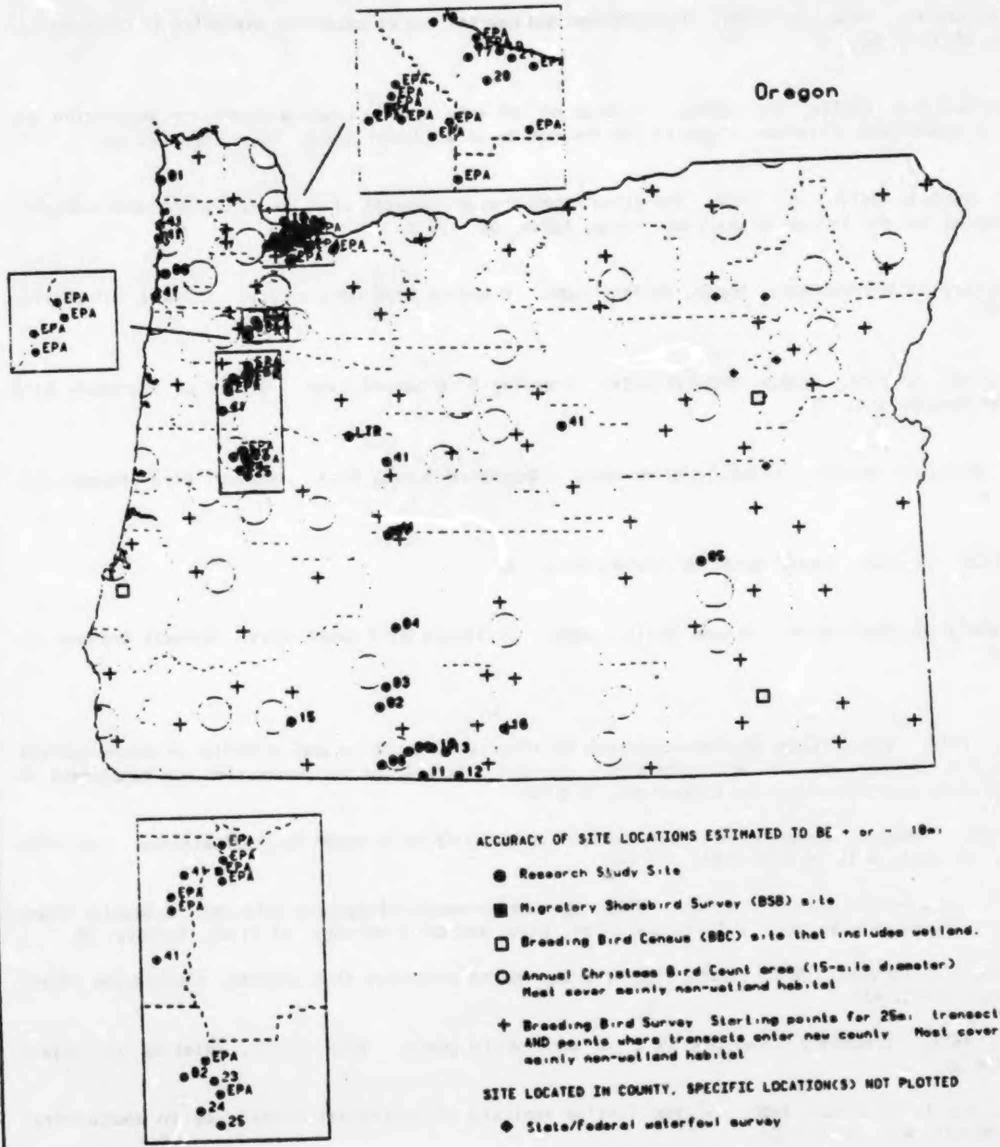
Inland Wetlands Having Biological Community Measurements



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

● Research Study Site

■ Migratory Shorebird Survey (BSB) site

☐ Breeding Bird Census (BBC) site that includes wetland.

○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat

+ Breeding Bird Survey Starting points for 25mi transects AND points where transects enter new county Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

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Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

July 1988

Data Compilation Paul Adamus and Robin Renteria

Cartography Jeff Irish

OREGON

Mapped

OR1
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OR2-4
Perdue, E.M., C.R. Lytle, M.S. Sweet, and J.W. Sweet. 1981. The Chemical and Biological Impact of Klamath Marsh on the Williamson River, Oregon. Environ. Sci. & Resour., Portland State Univ., Water Resour. Res. Inst., Oregon State Univ. WRRI-71, 199 pp. I A

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OR6-7
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OR17
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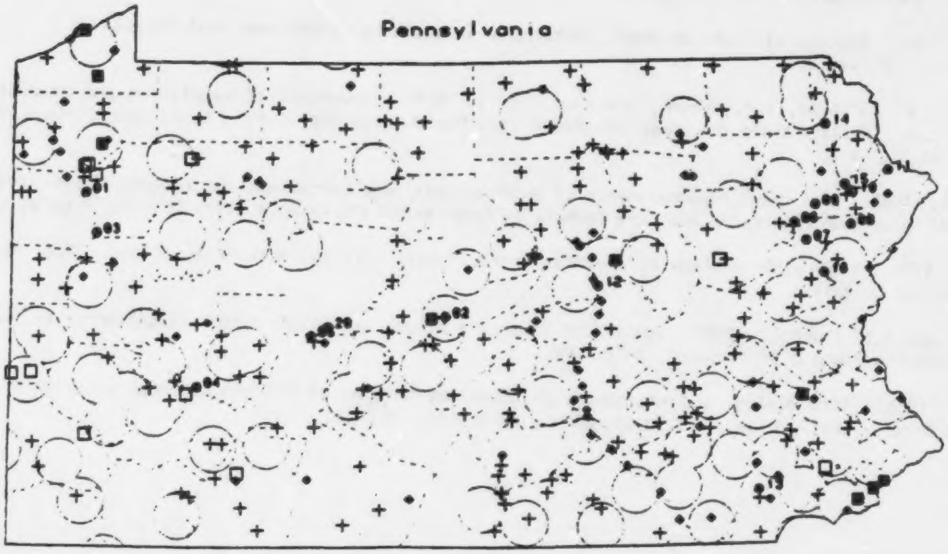
Inland Wetlands Having Biological Community Measurements



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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

PENNSYLVANIA

Mapped

PA1
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PA3-8
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PA14-16
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PA14-16
Bradt, P.T., J.L. Dudley, M.B. Berg, and D.S. Barrasso. 1986. Biology and chemistry of three Pennsylvania lakes: Responses to acid precipitation. Water, Air and Soil Poll. 30:505-513. AI I

PA20
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PABBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

PABBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

PABSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

PABW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

PACBC1-
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Not Mapped

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Sopper, W.E. and L.T. Kardos. 1973. Vegetation responses to irrigation with treated municipal wastewater. pp. 271-294. In: Recycling Treated Municipal Wastewater and Sludge Through Forest and Cropland, 271-294 pp., Pennsylvania State Univ. Press, Univ. Park, PA. P

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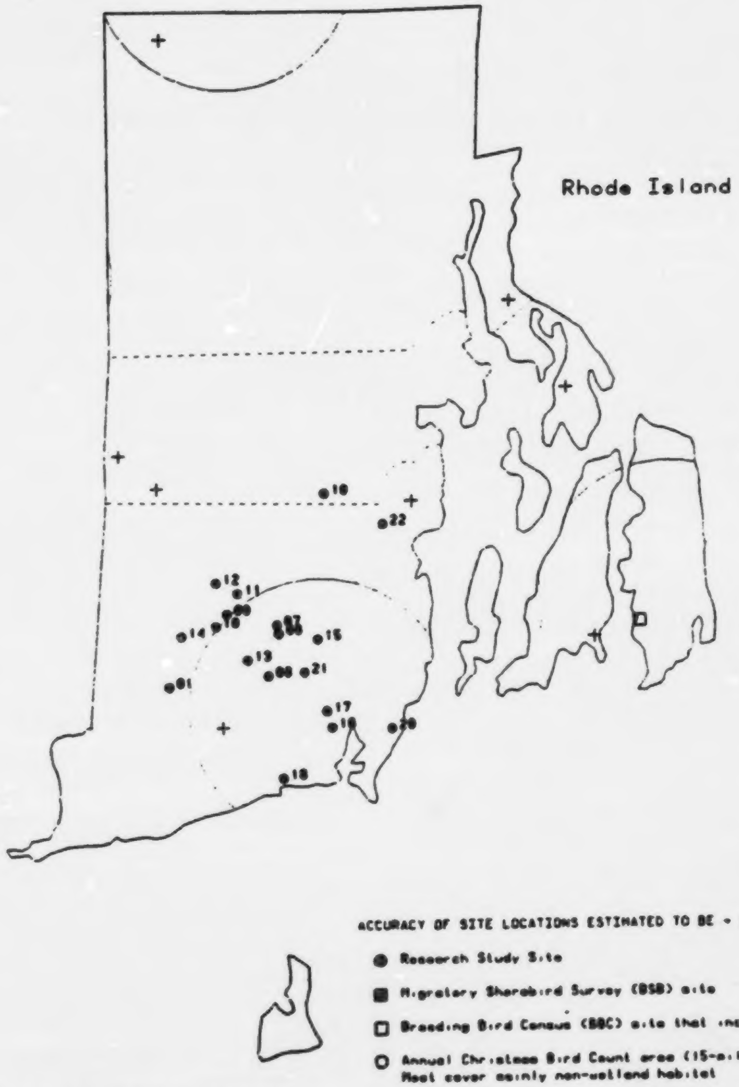
Inland Wetlands Having Biological Community Measurements



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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

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RHODE ISLAND

Mapped

RI1
Sheath, R.G., J.M. Burkholder, J.A. Hambrook, A.M. Hogeland, E. Hoy, M.E. Kane, M.O. Morison, A.D. Steinman, and K.L. Van Alstyne. 1986. Characteristics of softwater streams in Rhode Island. III. Distribution of macrophytic vegetation in a small drain. Hydrobiologia 140:183-191. PM R

RI6-22
Theses and Dissertations, Dept. Forest & Wildlife Management, Univ. Rhode Island, Kingston, RI.

RIBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

RIBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

RIBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

RICBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

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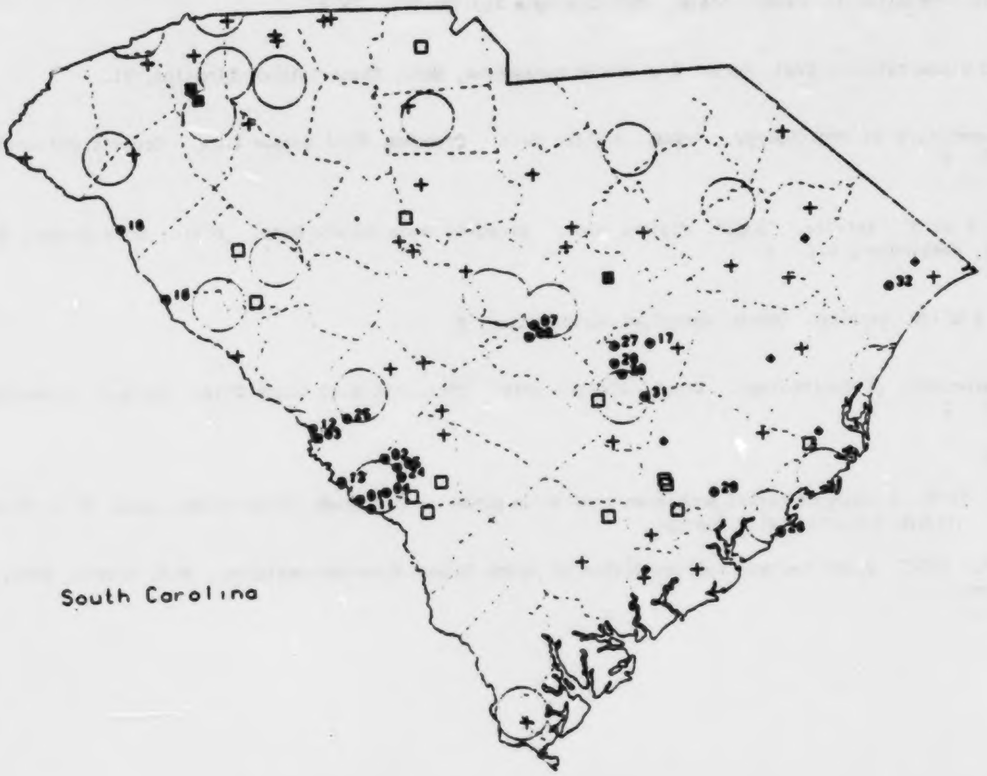
Inland Wetlands Having Biological Community Measurements


South Carolina
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ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi
● Research Study Site
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USEPA Environmental Research Laboratory, Corvallis, Oregon

July 1988

Data Compilation Paul Adamus and Robin Rantoria Cartography Jeff Irish

SOUTH CAROLINA

Mapped

SC1
Scott, M.L., R.R. Sharitz, and L.C. Lee. 1985. Disturbance in a cypress-tupelo wetland: An interaction between thermal loading and hydrology. Wetlands 5:53-68. I PW

SC2
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SC2
Sharitz, R.R. and L.C. Lee. 1985. Limits on regeneration processes in southeastern riverine wetlands. pp. 139-143 In: Riparian Ecosystems and Their Management: Reconciling Conflicting Uses. Gen. Tech. Rep. RM-120, USDA Forest Serv., Fort Collins, CO. PW

SC2
Sharitz, R.R., J.E. Irwin, and E.J. Christy. 1974. Vegetation of swamps receiving reactor effluents. Oikos 25:7-13. P I

SC2
Congdon, J.D., J.L. Greene, and J.W. Gibbons. 1986. Biomass of freshwater turtles: A geographic comparison. Amer. Midl. Nat. 115(1):165-173. H

SC2
Gibbons, J.W. and D.H. Bennett. 1974. Determination of anuran terrestrial activity patterns by a drift fence method. Copeia 1:236-243. H

SC3
Sheldon R.B. and C.W. Boylen. 1975. Factors affecting the contribution by epiphytic algae to the primary productivity of an oligotrophic freshwater lake. Appl. Microbiol. 30(4):657-667. A

SC4
Christensen, E.J., J.R. Jensen, E.W. Ramsey, and H.E. Mackey, Jr. 1988. Aircraft MSS data registration and
International J. Remote Sensing 9(1):23-38. RS
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SC5
Dunn, C.P. and M.L. Scott. 1987. Response of wetland herbaceous communities to gradients of light and substrate following disturbance by thermal pollution. Vegetatio 70:119-124. P I

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SC11
Paller, M.H. 1987. Distribution of larval fish macrophyte beds and open channels in a southeastern floodplain swamp. J. Freshw. Ecol. 4(2):191-200. F

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Thorp, J.H., E.M. McEwan, M.F. Flynn, and F.R. Hauer. 1985. Invertebrate colonization of submerged wood in a cypress-tupelo swamp and blackwater stream. Amer. Midl. Nat. 113(1):56-68. AI

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McArthur, J.V., L.G. Leff, D.A. Kovacic, and J. Jaroscak. 1986. Green leaf decomposition in coastal plain streams. J. Freshw. Ecol. 3(4):553-559. D

SC15
Pechmann, J.H.K., D.E. Scott, J.W. Gibbons, and R.D. Semlitsch. 1988. Influence of wetland hydroperiod on diversity and abundance of metamorphosing juvenile amphibians. Wetland Ecol. Manage. 1(1):3-11. H

SC15
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SC16
James, W.F., R.H. Kennedy, W.E. Shain, and R.K. Myers. 1988. Leaf litter breakdown in a recently impounded reservoir. Water Res. Bull. 24(4):831-837. D

SC17
U.S. Environmental Protection Agency. 1983. Hydrographic, Water Quality and Biological Studies of Freshwater Canal Systems, South Carolina, Mississippi, and Florida. Environ. Protection Agency, Environ. Serv. Div., Athens, GA. AI

SC18-23
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SC24
Schalles, J.F. and D.J. Shure. 1989. Hydrology, community structure, and productivity patterns of a dystrophic Carolina bay wetland. Ecol. Monogr. 59(4):365-385.

SC24
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SC25
Gibbons, J.W., J.L. Greene, and J.D. Congdon. 1983. Drought-related responces of aquatic turtle populations. J. Herpetology 17(3):242-246. H

SC26
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SC27
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SC28
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SC29
Homer, M.L. and J.B. Williams. 1986. The Effects of Aquatic Macrophyte Control on Fish Populations Inhabiting an Abandoned Rice Field in the Upper Cooper River, South Carolina. Dept. of Environ. Health Sci., Univ. of South Carolina, Columbia, SC. 170 pp. PM F

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SC30
McIlvaine, C.N. 1986. Seasonal abundance and diversity of zooplankton in upper Lake Marion, South Carolina. M.S. Thesis, Dept. of Environ. Health Sci., Univ. of South Carolina, Columbia, SC. 73 pp. AI

SC31
Welch, R. and M.M. Remillard. 1986. Aquatic Macrophyte Distributions in Lake Marion, South Carolina: June and September, 1985. Lab. for Remote Sensing and Mapping Sci., Dept. of Geography, Univ. of Georgia, Athens, GA. 15 pp. PM

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Welch, R., S.S. Fung, and M.M. Remillard. 1985. Aquatic Macrophyte Distribution in Lake Marion, South Carolina: 1983-1984. Lab. for Remote Sensing and Mapping Sci., Dept. of Geography, Univ. of Georgia, Athens, GA. 18 pp. PM

SC31
Welch, R., S.S. Fung, and M.M. Remillard. 1986. Changes in the Distribution of Aquatic Macrophytes: Lake Marion, South Carolina: 1972-1984.Lab. for Remote Sensing and Mapping Sci., Dept. of Geography, Univ. of Georgia, Athens, GA. 16 pp. PM

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CH2M Hill. Unpub. Central Slough Pilot Study. Grand Strand Water & Sewer Auth., Central Wastewater Treatment Plant Wetlands Discharge, Charleston, SC. P

SCBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

SCBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

SCBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

SCBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

SCCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

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Taylor, B.E., D.L. Mahoney, and R.A. Estes. 1989. Zooplankton production in a Carolina Bay. pp. 425-436 In: R.R. Sharitz and J.W. Gibbons (eds.). Freshwater Wetlands and Wildlife, Proceedings of a Symposium. CONF-8603101 (NTIS No. DE90005384). U.S. Dept. Energy, Washington, D.C.

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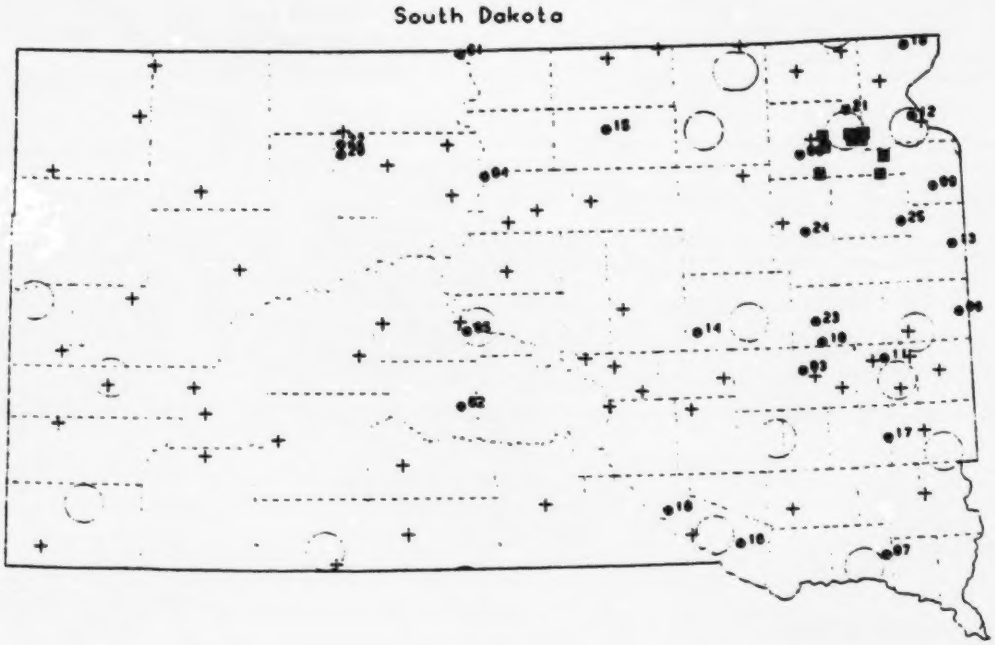
Inland Wetlands Having Biological Community Measurements



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi.

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SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

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SOUTH DAKOTA

Mapped

SD1
Duebbert, H.F. and A.M. Frank. 1984. Value of prairie wetlands to duck broods. Wildl. Soc. Bull. 12:27-34.

SD2
Rumble, M.A., and L.D. Flake. 1983. Management considerations to enhance use of stock ponds by waterfowl broods. J. Range. Manage. 36(6):691-694. B

SD3
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SD4
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SD5
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SD15-18
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SD19
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SD20
Hawkes, C.L. 1979. Aquatic habitat of coal and bentonite clay strip mine ponds in the northern Great Plains. Ecol. Coal Res. Dev.2:609-614. I P

SD21
Klett, A.T., T.L. Shaffer, and D.H. Johnson. 1988. Duck nest success in the Prairie Pothole region. J. Wildl. Manage. 52(3):431-440. B R

SD22
Uresk, D.W. and K. Severson. 1988. Waterfowl and shorebird use of surface-mined and livestock water impoundments on the Northern Great Plains. Great Basin Nat.48(3):353-357. B

SD23-25
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SDBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

SDBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

SDBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

SOUTH DAKOTA (continued)

SDCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

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Inland Wetlands Having Biological Community Measurements

Tennessee

ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi.

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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1990

TENNESSEE

Mapped

TN1
Fowler, D.K., D.M. Hill, and L.J. Fowler. 1985. Colonization of coal surface mine sediment ponds in Southern Appalachia by aquatic organisms and breeding amphibians. pp. 261-285. Penn. St. Univ., University Park, PA. In: R.P. Brooks, D.E. Samuel, and J.B. Hill (eds.). Wetlands and Water Management on Mined Lands. Penn. St. Univ., University Park, PA. AI H I

TN2
Pierce, C.L., P.H. Crowley, and D.M. Johnson. 1985. Behavior and ecological interactions of larval odonata. Ecol. 66(5):1504-1512. AI

TN3
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TN3
Cobb, S.P., C.H. Pennington, J.A. Baker, and J.E. Scott. 1984. Fishery and ecological investigations of main stem levee borrow pits along the lower Mississippi River. Mississippi R. Comm., Vicksburg, MS. 120 pp. F

TN5-39
Durham, D., E. Bridges, P. Hamel, S. Pearsall, L. Smith, and P. Somers. 1985. Conserving Natural Communities: Classification and Inventory. Tennessee Dept. of Cons., Ecol. Serv. Div., Nashville, TN.

TN40
Robinson, J.C. and D. Orr. 1988. A quantitative evaluation of moist soil management areas on Cross Creeks National Wildlife Refuge, Stewart County, Tennessee. Rep. 42515-03. US Fish & Wildl. Serv., Vicksburg, MS.

TN41-43
Young, R.C. and W.N. Dennis. 1983. Productivity of the Aquatic Macrophyte Community of the Holston River: Implications to Hypolimnetic Oxygen Depletions of Cherokee Reservoir. Tenn. Valley Authority. Div. of Air and Water Res.,TVA/ONR/WR-83/12. Muscle Shoals, AL.

TN44
Turner, L.J. and D.K. Fowler. 1981. Utilization of Surface Mine Ponds in East Tennessee by Breeding Amphibians. Off. Nat. Res., Div. of Land and Forest Res., TVA, Norris, TN.Contract # 14-16-0009-78-708. FWS/OBS-81/08. H

TN45
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TN46
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TNBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

TNBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

TNBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

TNCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

TENNESSEE (continued)

Not Mapped

Barstow, C.J. 1971. Impact of channelization on wetland habitat in the Obion-Forked Deer Basin, Tennessee. Trans. N. Amer. Wildl. Nat. Res. Conf. 36:362-376. I B

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Steenis, J.H. 1947. Recent changes in the marsh and aquatic plant status at Reelfoot Lake. J. Tennessee Acad. Sci. 22:22-27. RS P

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Webb, D.H. and A.L. Bates. 1989. The aquatic vascular flora and plant communities along rivers and reservoirs of the Tennessee river system. J. Tennessee Acad. of Sci. 64(3):197-203. PM P

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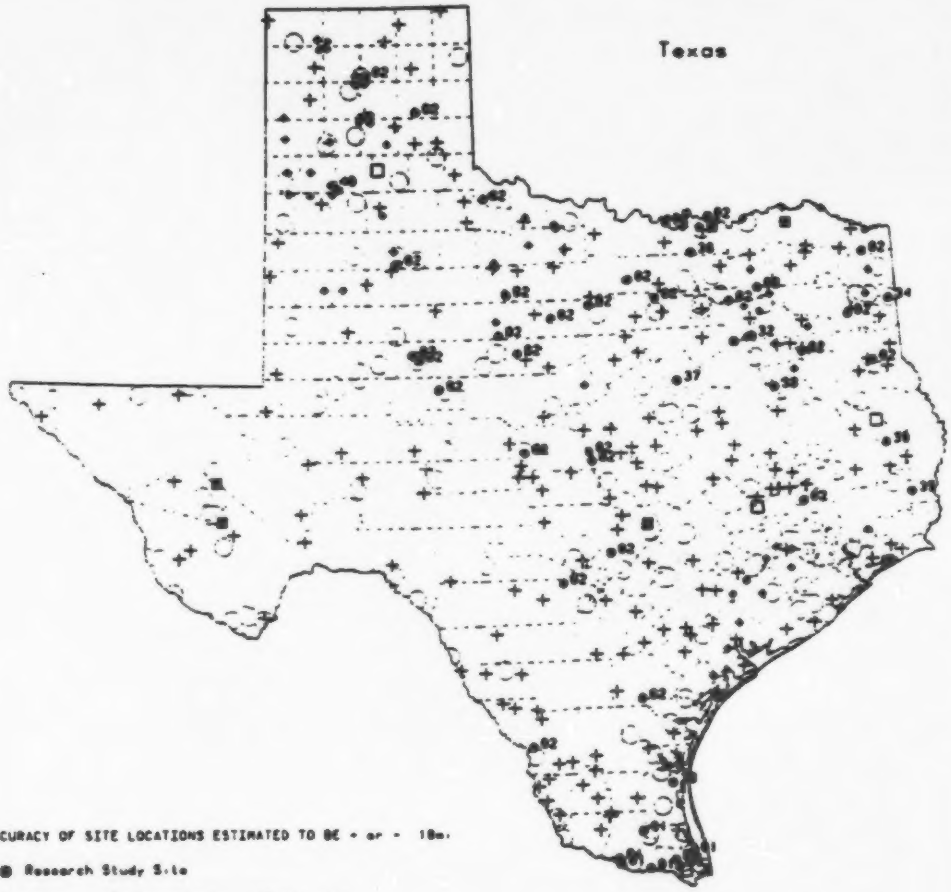
Inland Wetlands Having Biological Community Measurements



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USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation: Paul Adamus and Robin Renteria   Cartography: Jeff Irish   July 1988

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TEXAS

Mapped

TX1
Wells, F.C., G.A. Jackson, and W.J. Rogers. 1988. Reconnaissance Investigation of Water-Quality, Bottom Sediment, and Biota Associated with Irrigation Drainage in the Lower Rio Grange Valley and Laguna Atascosa National Wildlife Refuge, Texas, 1986-87. AI BA I

TX2
Durocher, P.D., W.C. Provine, and J.E. Kraai. 1984. Relationship between abundance of Largemouth Bass and submerged vegetation in Texas Reservoirs. N. Amer. J. Fish. Manage. 4:84-88. F R

TX34
Klimas, C.V. 1987. Baldcypress response to increased water levels, Caddo Lake, Louisiana-Texas. Wetlands 7:25-37. PW I

TX35
Streng, D.R., J.S. Glitzenstein, and P.A. Harcombe. 1989. Woody seedling dynamics in an East Texas floodplain forest. Ecol. Monogr. 59(2):177-204. PW

TX36
Institute of Applied Sciences, Univ. of North Texas. 1988. Pre-Impoundment Environmental Study of Ray Roberts Lake. Final Rep., Supplement to Design Memorandum No. 8. U.S. Army Corps of Engr., Fort Worth.

TX37
Slack, R.D. and L.E. Marcy. 1983. Pre-Impoundment Environmental Study of Aquilla Lake.Final Report. Supplement to Design Memorandum No. 9, U.S. Army Corps of Engr., Fort Worth.

TX37
Slack, R.D., B.R. Murphy, W.J. Spearman, and J. Hinson. 1989. Aquilla Lake Environmental Study (Year Five).Final Report. U.S. Army Corps of Engr., Fort Worth Dist., Supplement to Design Memorandum No. 9, Texas A & M Univ., Dept. of Wildl. & Fish. Sci.

TX37
Slack, R.D., Maceina, M.J., and M.D. Hoy. 1986. Post-Impoundment Environmental Study (Year-Two) of Aquilla Lake.Final Report. U.S. Army Corps of Engr., Fort Worth Dist., Supplement to Design Memorandum No. 9, Texas A & M Univ., Dept. of Wildl. & Fish. Sci.

TX38
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TX39
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TX40
Baldassarre, G.A. and E.G. Bolen. 1984. Field-feeding ecology of waterfowl on the southern High Plains of Texas. J. Wildl. Manage. 48:63-71. B

TX41
Hobaugh, W.C. and J.G. Teer. 1981. Waterfowl use characteristics of flood-prevention lakes in north-central Texas. J. Wildl. Manage. 45:16-26. B

TXBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

TXBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

TEXAS (continued)

TXBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

TXBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

TXCBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

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Allen, C.E. 1975. Bioeconomics and feeding habits of ducks in flooded bottomlands of eastern Texas. M.S. Thesis, Stephen F. Austin St. Univ., Nacogdoces, TX. 66 pp.

Bacak-Clements, P.M. 1988. A survey of avian use and the aquatic fauna of three ponds, in Willacy County, Texas. M.S. Thesis, Texas A&M Univ., College Station, TX. 124 pp.

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Becker, P.R. 1972. Secondary production of selected invertebrates in an ephemeral pond. Ph.D. Diss., Texas A&M Univ., College Station, TX. 154 pp.

Benson, D.J., L.C. Fitzpatrick, and W.D. Pearson. 1980. Production and energy flow in the benthic community of a Texas pond. Hydrobiologia 74:81-93. AI

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Campbell, J.M. 1983. Interpond and intrapond variation in populations of periphytic cladoceran microcrustacea. Ph.D. Diss., Texas A&M Univ., College Station, TX. 298 pp.

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Duncan, K.L. 1988. The effects of seismic exploration on the woody vegetation of the Big Thicket National Preserve. M.F. Thesis, Stephen F. Austin St. Univ., Nacogdoces, TX. 192 pp.

Evans, J.D. 1988. An ecological study of the crustacean community in a prairie temporary marsh in central Texas. M.S. Thesis, Stephen F. Austin St. Univ., Nacogdoces, TX. 124 pp.

TEXAS (continued)

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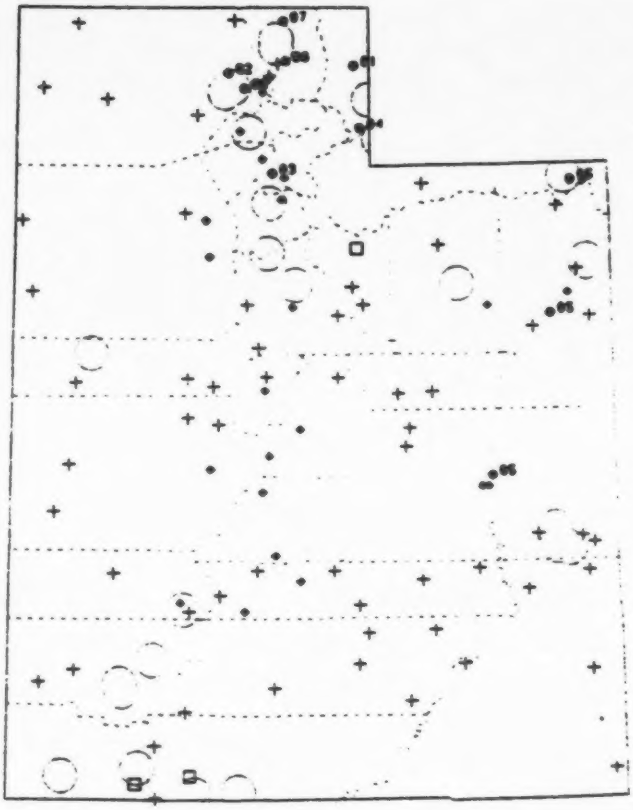
Inland Wetlands Having Biological Community Measurements



Utah

ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi.

● Research Study Site

■ Migratory Shorebird Survey (BSB) site

☐ Breeding Bird Census (BBC) site that includes wetland

○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat

+ Breeding Bird Survey Starting points for 25mi transects AND points where transects enter new county Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

◆ State/Federal waterfowl survey

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Sites are referenced by code number to the accompanying state bibliography

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UTAH

Mapped

UT1
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UT3
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UT5
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UT6
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UT7
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UT8
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UT8
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UTBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

UTBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

UTBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

UTCBC1-
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Not Mapped

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Inland Wetlands Having Biological Community Measurements

ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10mi

- ● Research Study Site
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- ☐ Breeding Bird Census (BBC) site that includes wetland
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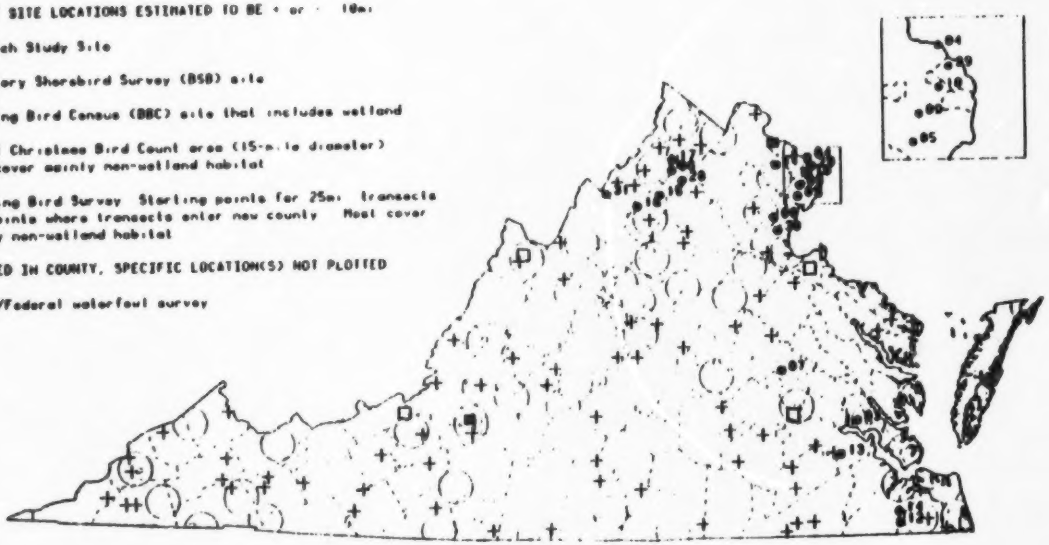
SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

- ◆ State/Federal waterfowl survey



Virginia

This map does NOT portray ALL wetland sampling sites
Emphasis is on sites where community-level data were
collected See chapter 1 for inclusion criteria

Sites are referenced by code number to the accompanying
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Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

VIRGINIA

Mapped

VA3
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VA4-6
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VA28-30
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Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

VABBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

VABSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

VABW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

VACBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Christmas Bird Count Data. Cornell University, Ithaca, NY. B

Not Mapped

Breidling, F.E., F.P. Day, Jr., and R.K. Rose. 1987. An evaluation of small rodents in four Dismal Swamp plant communities. Virginia J. Sci. 34(1):14-28. MA

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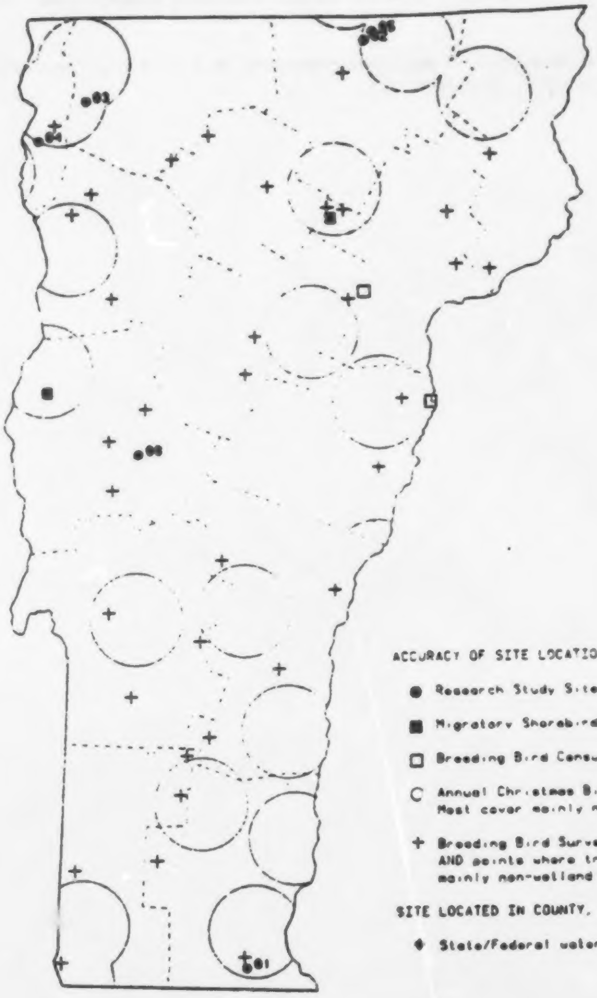
Inland Wetlands Having Biological Community Measurements



Vermont

ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10m.

- ● Research Study Site
- ■ Migratory Shorebird Survey (BSB) site
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- C Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat
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SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

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USEPA Environmental Research Laboratory, Corvallis, Oregon

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VERMONT

Mapped

VT2
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VT8
Dodge, W.E., E.E. Possardt, R.J. Reed, and W.P. MacConnell. 1976. Channelization Assessment, White River, Vermont: Remote Sensing, Benthos, and Wildlife. FWS/OBS-76-07. U.S. Fish & Wildl. Serv., Washington, D.C. 73 pp.

VTBBC1-
Cornell Laboratory of Ornithology. Unpub. digital data. Breeding Bird Census Data. Cornell University, Ithaca, NY. B

VTBBS1-
U.S. Fish & Wildl. Service. Unpub. digital data. Breeding Bird Survey Data. Office of Migratory Bird Management, Washington, D.C. B

VTBSB1-
International Shorebird Survey. Unpub. digital data. Shorebird Survey Data. Manomet Bird Observatory, Manomet, MA. B

VTBW1-
U.S. Fish & Wildl. Service. Unpub. Waterfowl Survey Data. B

VTCBC1-
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VERMONT (continued)

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Strimbeck, G.R. 1988. Fire, flood, and famine: pattern and process in a lakeside bog. M.S. Thesis, Botany Dept, Univ. Vermont, Burlington, VT. P

Vermont Agency of Natural Resources. 1990. The Lake Bomoseen Drawdown: An Evaluation of its Effect on Aquatic Plants, Wildlife, Fish, Invertebrates, and Recreational Uses. Waterbury, VT. 282 pp.

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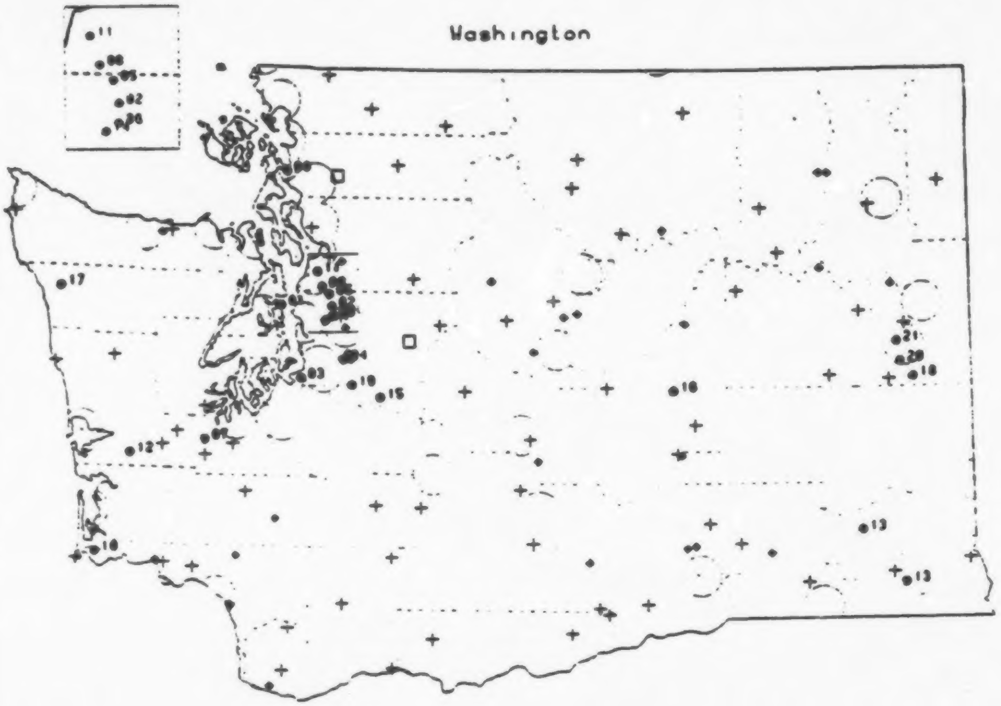

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WASHINGTON

Mapped

WA1-6
Milligan, D.A. 1985. The ecology of avian use of urban freshwater wetlands in King county, Washington. M.S. Thesis, Univ. Washington. B I

WA7
Meehan-Martin P.J. and D. Swanson. 1988. Pacific Avenue Interchange , SR 5. Wetland Monitoring Report. Washington State Dept. Trans. Environ. Unit, Olympia. Job # L-6941. 12 pp. P

WA8
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WA9
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WA10
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WA10
Verhalen, F.A., M.L. Gibbons, and W.H. Funk. 1985. Implications for control of Eurasian water milfoil in the Pend Oreille River. Lake and Reservoir Management - Practical Applications, 361 pp.

WA11
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WA12
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WA13
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WA18
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WABW1-
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WACBC1-
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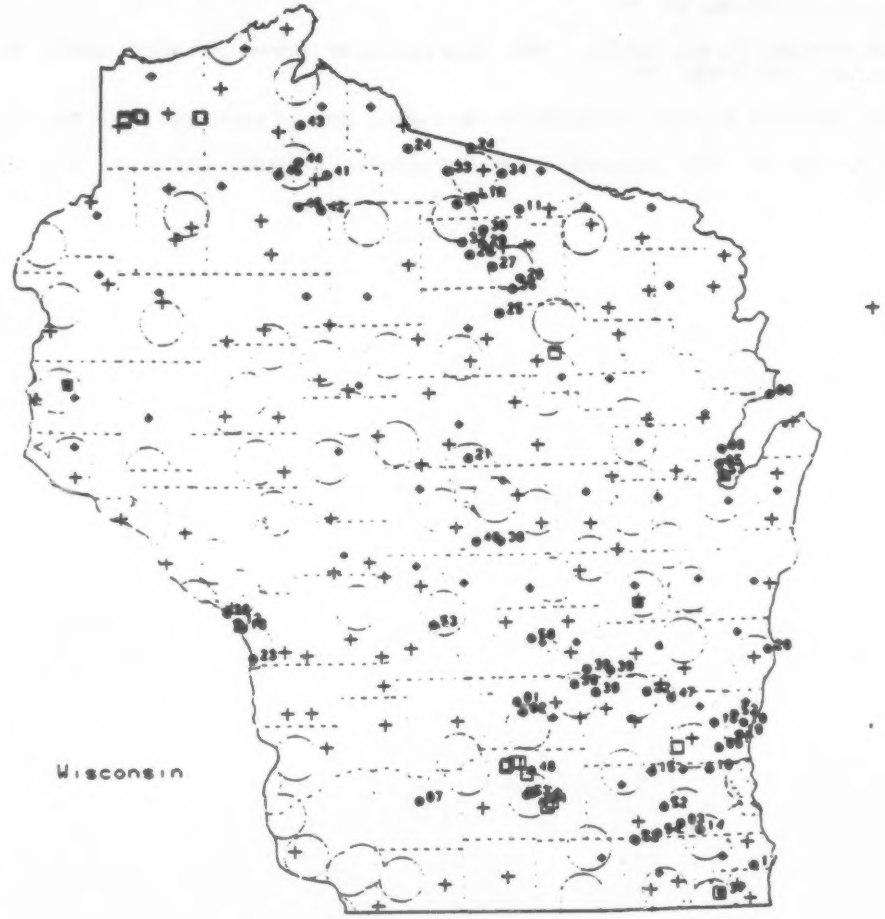
Inland Wetlands Having Biological Community Measurements



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10m.

- ● Research Study Site
- ■ Migratory Shorebird Survey (BSB) site
- ☐ Breeding Bird Census (BBC) site that includes wetland
- ○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat.
- + Breeding Bird Survey Starting points for 25mi transects AND points where transects enter new county Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

- ◆ State/Federal waterfowl survey

This map does NOT portray ALL wetland sampling sites Emphasis is on sites where community-level data were collected See chapter 1 for inclusion criteria.

Sites are referenced by code number to the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

WISCONSIN

Mapped

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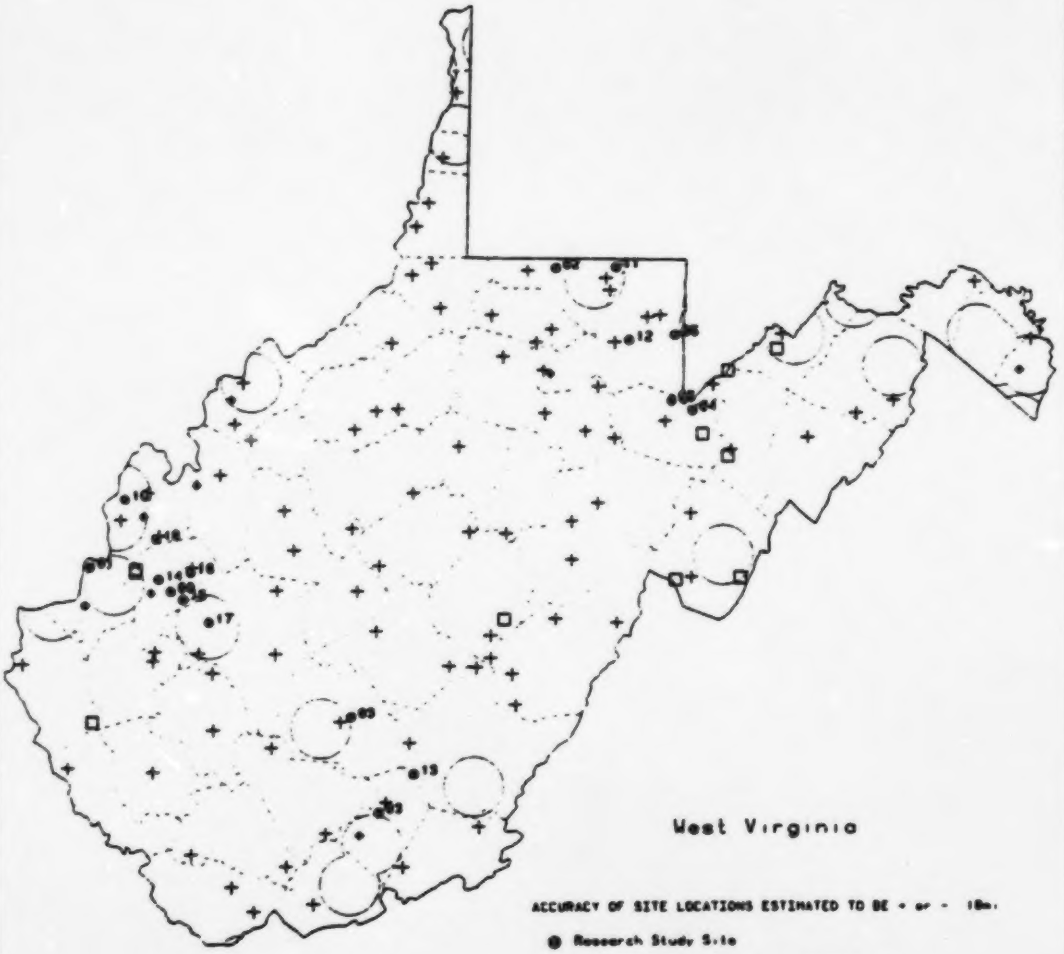
Inland Wetlands Having Biological Community Measurements



West Virginia

ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10m.

● Research Study Site

■ Migratory Shorebird Survey (MSS) site

□ Breeding Bird Census (BBC) site that includes wetland

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Most cover mainly non-wetland habitat

+ Breeding Bird Survey Starting points for 25mi. transects
AND points where transects enter new county Most cover
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SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

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This map does NOT portray ALL wetland sampling sites
Emphasis is on sites where community-level data were
collected See chapter 1 for inclusion criteria

Sites are referenced by code number to the accompanying
state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria   Cartography Jeff Irish   July 1988

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best available copy.

WEST VIRGINIA

Mapped

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WV4
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WV9-10
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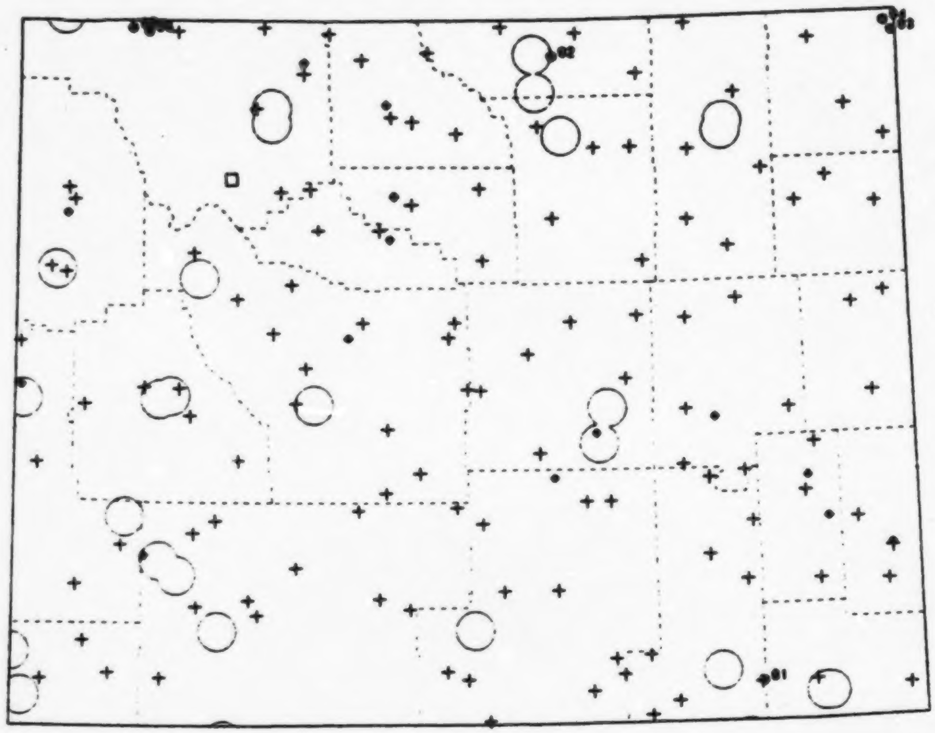
Inland Wetlands Having Biological Community Measurements

Wyoming



ACCURACY OF SITE LOCATIONS ESTIMATED TO BE + or - 10m.

- ● Research Study Site
- ■ Migratory Shorebird Survey (MSB) site
- □ Breeding Bird Census (BBC) site that includes wetland
- ○ Annual Christmas Bird Count area (15-mile diameter) Most cover mainly non-wetland habitat
- + Breeding Bird Survey Starting points for 25mi. transects AND points where transects enter new county Most cover mainly non-wetland habitat

SITE LOCATED IN COUNTY, SPECIFIC LOCATION(S) NOT PLOTTED

- ◆ State/Federal waterfowl survey

This map does NOT portray ALL wetland sampling sites Emphasis is on sites where community-level data were collected See chapter 1 for inclusion criteria

Sites are referenced by code number in the accompanying state bibliography

USEPA Environmental Research Laboratory, Corvallis, Oregon

Data Compilation Paul Adamus and Robin Renteria Cartography Jeff Irish July 1988

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WYOMING

Mapped

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WYBBC1-
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WYBBS1-
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